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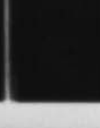
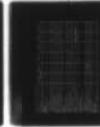
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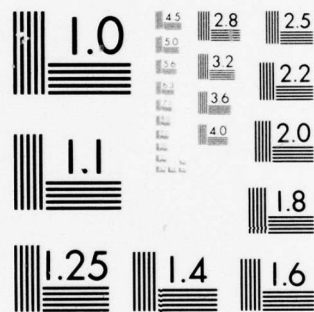
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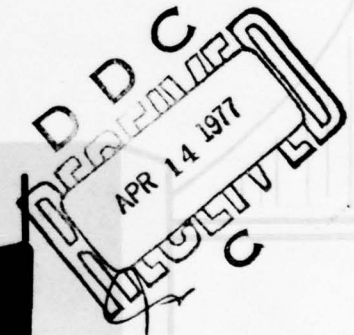
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DREDGE DISPOSAL STUDY

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SAN FRANCISCO BAY AND ESTUARY



APPENDIX I

POLLUTANT AVAILABILITY

OCTOBER 1975

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⑥ DREDGE DISPOSAL STUDY, SAN FRANCISCO BAY AND ESTUARY.
APPENDIX I.
POLLUTANT AVAILABILITY STUDY.

⑨ Final rept.

⑫ 12329p.

⑪ OCT 1975

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FOREWORD

In April 1972, the San Francisco District of the United States Army Corps of Engineers initiated a three and one-half year \$3 million study to quantify the impact of dredging and dredged material disposal operations on the San Francisco Bay and Estuarine environment. The study is generating factual data, based on field and laboratory studies needed for the Federal, State and local regulatory agencies to evaluate present dredging policies and alternative disposal methods.

The study is set up to isolate the questions regarding the environmental impact of dredging operations and to provide answers at the earliest date. The study is organized to investigate (a) the factors associated with dredging and the present system of aquatic disposal in the Bay, (b) the condition of the pollutants (biogeochemical), (c) alternative disposal methods, and (d) dredging technology. The study elements are intended first, to identify the problems associated with dredging and disposal operations and, second, to address the identified problems in terms of mitigation and/or enhancement. The division into separate but inter-related study elements provides a greater degree of expertise and flexibility in the Study.

This report presents the findings of Appendix I, Pollutant Availability. The final report of the Dredge Disposal Study will integrate the information contained within this Appendix and information from other appendices.

The following is an index of appendices to be published in the Dredge Disposal Study:

<u>APPENDIX</u>	<u>REPORT</u>	<u>DATE PUBLISHED</u>
	FINAL REPORT	
A	Main Ship Channel (San Francisco Bar)	June 1974
B	Pollutant Distribution	
C	Water Column (Water Column-Oxygen Sag)	
D	Biological Community	August 1975
E	Material Release	
F	Crystalline Matrix	July 1975
G	Physical Impact	July 1975
H	Pollutant Uptake	September 1975
I	Pollutant Availability	October 1975
J	Land Disposal	October 1974
K	Marsh Development	
L	Ocean Disposal	September 1975
M	Dredging Technology	September 1975

CONVERSION FACTORS

If conversion from the Metric to the British system is necessary, the following factors apply:

LENGTH

1 kilometer (km) = 10^3 meters = 0.621 statute miles = 0.540 nautical miles
1 meter (m) = 10^2 centimeters = 39.4 inches = 3.28 feet = 1.09 yards = 0.547 fathoms
1 centimeter (cm) = 10 millimeters (mm) = 0.394 inches = 10^4 microns (μ)
1 micron (μ) = 10^{-3} millimeters = 0.000394 inches

AREA

1 square centimeter (cm²) = 0.155 square inches
1 square meter (m²) = 10.7 square feet
1 square kilometer (km²) = 0.386 square statute miles = 0.292 square nautical miles

VOLUME

1 cubic kilometer (km³) = 10^9 cubic meters = 10^{15} cubic centimeters = 0.24 cubic statute miles
1 cubic meter (m³) = 10^6 cubic centimeters = 10^3 liters = 35.3 cubic feet = 264 U.S. gallons = 1.308 cubic yards
1 liter = 10^3 cubic centimeters = 1.06 quarts = 0.264 U.S. gallons
1 cubic centimeter (cm³) = 0.061 cubic inches

MASS

1 metric ton = 10^6 grams = 2,205 pounds
1 kilogram (kg) = 10^3 grams = 2.205 pounds
1 gr (g) = 0.035 ounce

SPEED

1 knot (nautical mile per hour) = 1.15 statute miles per hour = 0.51 meter per second
1 meter per second (m/sec) = 2.24 statute miles per hour = 1.94 knots
1 centimeter per second (cm/sec) = 1.97 feet per second

TEMPERATURE

Conversion Formulas

$$^{\circ}\text{C} = \frac{^{\circ}\text{F} - 32}{1.8}$$

$$^{\circ}\text{F} = 1.8(^{\circ}\text{C}) + 32$$

ERRATA

to

POLLUTANT AVAILABILITY STUDY

*V.A. Anderlini, J.W. Chapman, A.S. Newton, R.W. Risebrough, B.E. Cole,
B.W. de Lappe, D.C. Girvin, A.T. Hodgson, S.J. McCormick,
L. Nelbach, R.K. Okazaki, M.H. Panietz and T.T. Schmidt*

- (1) page 6, fourth paragraph, third line from bottom:
change "300 meters" to "200 meters"
- (2) page 6, fifth paragraph, seventh line:
change "According to his calculations, a 1 meter high mound of
spoil would disperse between 100 and 500 m." to "According to
his calculations, a 1 meter high mound of spoil would have a
diameter between 100 and 500 m."
- (3) page 9, second line from bottom:
"current" should be "velocity"
- (4) page 14, third paragraph, third line:
"sample (TO.08)" should be "sample T-.08"
- (5) page 16, fifth paragraph, second line:
"stainless stell pestle" should be "stainless steel pestle"
- (6) page 28, first paragraph, sixth line:
"along" should be "alone"
- (7) page 102, first paragraph, fourth line from bottom:
"currents or by re-adsorption" should be "currents and/or
adsorption"
- (8) page 102, second paragraph, tenth line:
"significant increase in trace element concentrations" should
be "significant increases in any single trace element
concentration was"
- (9) page 102, fourth line from bottom:
"occurred directly" should be "occurred once directly"
- (10) page 103, third line:
starting with "unbiased" delete the remainder of the sentence
and insert "spoiling created little effect on overall settling"

rates and average suspended particulate loads."

- (11) page 103, fourth paragraph:
delete entire last sentence and insert "Only increases of very short duration and/or below detection limits of the methods applied could have been unobserved."
- (12) page 104, eleventh line from bottom:
delete "(U.S. Army Corps of Engineers)" and insert "(Anonymous, 1975)"
- (13) page 104, fifth line from bottom:
delete "(U.S. Army Corps of Engineers)" and insert "(Anonymous, 1973)"
- (14) page 106, first line:
delete "are valid within an order of magnitude since" and insert "of 1165 km²." Capitalize the "o" in "our" to begin the next sentence.

(For further information on sediment loads in San Francisco Bay, see:
Conomos, T.J. and D.H. Peterson. 1975. Suspended particulate transport and circulation in San Francisco Bay: an overview. Proc. Third Meeting Int. Estuar. Res. Fed., Oct. 75, Galveston, Texas.)
- (15) page 108, tenth line from bottom:
"class was initiated" should be "class were initiated"
- (16) page 109, thirteenth line from bottom:
add "general" to the end of the line
- (17) page 112, conclusion #2, first line:
"concentrations remained stable" should be "concentrations remained relatively stable"
- (18) page 112, conclusion #6, fourth line:
delete "estuarine"
- (19) page 113, conclusion #5, fifth line:
"currents or by" should be "currents and/or by"
- (20) page 116:
insert below the citation "Anonymous, 1971":

1973. Oakland Inner Harbor analysis of sediment samples. South Pacific Div. Lab., Sausalito, Calif., U.S. Army Corps Engin. 5 pp.

1975. Final composite environmental statement: Maintenance dredging: Existing navigation projects, San Francisco Bay region California. U.S. Army Corps Engin., San Francisco Vol. 1, Sect. I, 73 pp.

POLLUTANT AVAILABILITY STUDY

SUBMITTED TO

U.S. ARMY CORPS OF ENGINEERS

BY

LAWRENCE BERKELEY LABORATORY
ENERGY & ENVIRONMENT DIVISION¹

AND

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ABSTRACT

An integrated investigation of the effects of a dredge hopper disposal operation on pollutant availability to local invertebrate fauna and of the pathways (water, sediment, and suspended particulates) by which pollutants may be accumulated by invertebrates was undertaken in San Francisco Bay. Ten thousand m³ of polluted sediments from Oakland Inner Harbor were dumped at an experimental Spoil Site. Twelve trace elements (Ag, As, Cd, Cr, Cu, Fe, Hg, Mn, Ni, Pb, Se, and Zn) and the chlorinated hydrocarbon compounds of the PCB and DDT groups were monitored in selected benthic invertebrates, mussels transplanted to the Spoil Site, sediments, settled and suspended particulates, and water before, during, and after a 42-hour disposal operation.

The impact of such a spoiling operation in relation to the input of pollutants from a major point source, the East Bay Municipal Utility District's San Francisco Bay sewage outfall, was determined. Stations were established adjacent to the EBMUD outfall and pollutants were monitored in benthic invertebrates, transplanted mussels, sediment, and settled particulates at this site during the period of study.

Water parameters (salinity, temperature, pH, nitrate, and ammonia nitrogen and dissolved oxygen) were monitored before, during, and after the experimental spoiling to provide information on the physical-chemical conditions which may accompany or be associated with fluctuations in pollutant concentrations.

The pathways by which pollutants may be accumulated by invertebrates were examined in laboratory studies and in a "Field Laboratory" experiment in which selected invertebrate species were exposed in situ to altered concentrations of suspended particulates.

Under the conditions described in this report, the experimental spoiling operation did not appreciably affect trace element concentrations in the invertebrates examined. Chlorinated hydrocarbon concentrations in Spoil Site mussels fluctuated slightly during the period of study and only p,p'-DDE levels in mussels appear to have been affected by the spoiling operation. DDE levels decreased significantly more in mussels at stations outside of the immediate spoil zone and at other central bay locations than in mussels immediately adjacent to the disposal center. This effect lasted for less than one month. Copper and Fe concentrations in surface sediments were found to be higher at stations within the disposal area than at stations removed from the impact center after spoiling operations. Short term increases in dissolved Cd, Cu, and Pb concentrations and in chlorinated hydrocarbon levels were observed in samples collected from within the spoil plume immediately after each of the experimental disposals. These increases were significantly greater than pre- or post-spoiling natural fluctuations

of these pollutants. Higher metal concentrations persisted for less than 1.5 hours; chlorinated hydrocarbon levels returned to pre-spoil concentrations in less than 0.5 hours.

The amount of trace elements redistributed annually by all dredging activities is much greater than the annual input from the EBMUD outfall, but is almost inconsequential in relation to element redistribution by settling particulates.

Dredging and spoiling activities were found to redistribute polluted sediments without, under the conditions of the experiment conducted, resulting in increased pollutant availability. The EBMUD outfall was found to be a source of both trace element and chlorinated hydrocarbon inputs into the San Francisco Bay estuary.

Water quality conditions as defined by the water chemistry and suspended and settled particulate load data remained stable, without significant longer-term fluctuations, during the period of study.

Dissolved Hg was accumulated directly from water in laboratory studies by the three invertebrate species Macoma nasuta, Pectinaria californiensis and Mytilus edulis. Dissolved Pb and Cd were accumulated from water by M. nasuta and M. edulis, but little, if any, accumulation of Pb and no accumulation of Cd was observed in P. californiensis. A 50% reduction in the total suspended particulate load in the "Field Laboratory" study did not affect trace element concentrations in the invertebrates examined.

Fluctuations in the concentrations of the twelve elements were highly correlated with each other in sediment, invertebrates, suspended and settled particulates, although element levels within any one component were not correlated with element levels in any other component. The high correlation among trace elements within each component suggests that only one or a few parameters may control trace element fluxes in San Francisco Bay.

ACKNOWLEDGEMENTS

This study was funded by the San Francisco District, U.S. Army Corps of Engineers. We would like to thank Mr. John Sustar and Mr. Thomas Wakeman of the Army Corps for their advice and encouragement throughout the period of this study.

We also thank Dr. J. M. Hollander of the Lawrence Berkeley Laboratory (LBL) and Dr. Cadet Hand of the Bodega Marine Laboratory and their staffs for their consideration and help. We are especially indebted to Lieutenant-Commander LCDR W. Caster and Mr. Ward, Chief Polease, Chief Chung, and the men and women of the U.S. Coast Guard Station, Yerba Buena Island. Their help, advice, and good wishes enabled us to complete many of our more difficult collections. We want to express our gratitude to Mr. Aldo Sciamanna at LBL for his tireless efforts, ingenuity and enthusiasm for this project. We are also grateful to Ms. Roberta Garrett, Ms. Katherine Breyer, Ms. Mary Anne Benton, Ms. Mary Ann Terry, Ms. Phyllis Pollack and Ms. Carolyn Heath for their invaluable help in the preparation and analysis of the numerous samples analyzed during this study. We are grateful to Mr. Robert Giaugue and Dr. Tetsu Hadeishi for their advice and for allowing us to use their analytical facilities at LBL. We express our appreciation to the many people at LBL whose help, encouragement and friendship made our work possible.

We would also like to thank the captains and crews of the CHESTER HARDING, GRISSLY, and RACCOON for their cooperation, Mr. R. R. Ahern of the Western Pacific Railroad for his kindness and help, and Mr. Greg Brazil and the Berkeley Sea Scouts for their aid and endurance.

Lastly we thank Ms. Connie Serriere at LBL for her help in organizing and preparing this manuscript; Ms. Leslie Power and Ms. Lorraine Andrade of the Bodega Marine Laboratory typed the final drafts.

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INTRODUCTION

This report, Appendix I of the report of the three-year Corps of Engineers Dredge Disposal Study, describes a six-month investigation of the availability of pollutants to invertebrates in San Francisco Bay in relation to Corps of Engineers dredge disposal operations.

Availability is defined in this study as a measure of the uptake and accumulation by estuarine invertebrates of pollutants present in such input sources as dredge material and waste water effluents. Pollutants in these sources are therefore not considered "available" if input does not result in an elevation of pollutant levels in these invertebrates.

Controversy exists whether pollutants present in sediments are released during dredging and spoiling operations. Durant and Reimold (1972) did not observe increased toxaphene concentrations in oysters and sediments during a dredging operation in a contaminated Georgia estuary. In laboratory studies, Jernelov and Lann (1973) observed increased methylation of Hg in resuspended sediments, whereas they did not observe increases in the methyl mercury concentrations of fish during the dredging of mercury-contaminated lake sediments. They concluded that biological methylation of mercury is likely to occur during dredging operations and that minimal sediment resuspension was desirable. From a preliminary laboratory study of San Francisco Bay sediments, Gustafson (1972) concluded that resuspension of dredged sediments would result in decreased concentrations of soluble heavy metals and that no release would occur. However, the laboratory investigations of Serne and Mercer (1975) on trace element sorption and desorption in San Francisco Bay sediments, indicate that cadmium and possibly other elements could be released during the dredging-spoiling process.

A number of large surveys such as those conducted by Brehmer (1967) in Chesapeake Bay and by Pearce (1970) in New York Bight have investigated the general environmental effects of dredging and spoiling activities. Other studies have dealt with specific physical or chemical effects resulting from dredging operations. Kaplan *et al.* (1974) and May (1973) observed hydrological changes at disposal sites, and Pfitzenmeyer (1970) and Halcrow *et al.* (1973) examined substrate alteration as a result of spoiling operations. The effects of dredging and spoiling on turbidity were investigated by Flemer *et al.* (1968), Gordon (1974) and Rose (1973), while Brown and Clark (1968) and Wakeman *et al.* (1974) monitored concentrations of dissolved oxygen during dredging and spoiling activities.

Some studies have dealt with the geochemical changes that occur in sediments and associated interstitial waters. Presley *et al.* (1972) studied the distribution of trace elements in sediments and associated interstitial waters. Olausson (1970) examined water-

sediment exchange and recycling of trace elements and chlorinated hydrocarbons in estuarine basins; Windom (1973) investigated changes in heavy metal concentrations of sediments resulting from maintenance dredging of Mobile Bay Ship Channel.

Other studies related to the availability of pollutants include the early work of Krauskopf (1956) in which the factors controlling trace element concentrations in seawater were examined and the work of Brooks *et al.* (1968) in which factors controlling trace element levels in interstitial waters of marine sediments were studied. Mortimer (1971) speculated on the various factors which may control chemical exchanges between water and sediments in the Great Lakes and Duchart *et al.* (1973) reported trace element levels in the interstitial waters of marine sediments from Scotland. The work of Cross *et al.* (1970) included a study not only of the partitioning of metals between sediments and water but also of partitioning among sediments, water, and organisms.

Sediments have been shown to be major sinks in the marine and estuarine environment for a number of toxic metals. Concentrations of lead and copper and of mercury in San Francisco Bay sediments were reported by Peterson *et al.* (1972) and McColloch *et al.* (1971), respectively. Moyer and Budinger (1974) determined the cadmium concentration of near shore sediments from San Francisco Bay; Anderlini *et al.* (1975) presented the concentrations of nine trace elements observed in sediments adjacent to a dredged ship channel in San Francisco Bay. Chow *et al.* (1973) have demonstrated the accumulation of anthropogenic lead in deep sea sediments off Southern California. Carmody *et al.* (1973) observed 10-100 times higher concentrations of chromium, copper, lead, nickel and zinc in sediments near waste disposal areas than in sediments from adjacent areas of New York Bight. Holmes *et al.* (1974) related high concentrations of cadmium and zinc in the sediments of a marine estuary to heavy industrial discharges.

Sediments have also been shown to be sinks for chlorinated hydrocarbon compounds. Albone *et al.* (1972) showed that sediments in the Severn Estuary were a major sink for DDT compounds and Reimold and Durant (1974) found sediments to be a sink for toxaphene from a nearby toxaphene manufacturing plant.

The factors considered in this study to influence availability of pollutants to invertebrates in San Francisco Bay consisted of: (1) the magnitude of the various inputs; (2) the rates of degradation or removal; (3) the physical and chemical characteristics of the receiving waters, including suspended particulates; (4) the physical and chemical characteristics of the respective pollutants which determine how each is distributed among sediments, suspended particulates, water, and organisms; and (5) the physiological state of the organisms exposed to the pollutants.

Water, suspended particulates, and sediment constitute major routes by which pollutants may enter estuarine invertebrates. Concentrations in one component are dependent on concentrations in the other components and pollutant movement along or among these pathways involves complex, physical-chemical equilibrium processes that are poorly understood.

To provide data on the movement of pollutants along each of these routes and to determine the effects of dredge material disposal on the availability of these pollutants to invertebrates, 10,000 m³ of polluted sediment were dredged from above the Posey Tube in Oakland Inner Harbor and dumped at an experimental disposal site east of Angel Island in central San Francisco Bay (Figure 1). Trace element concentrations were monitored in sediments, water, suspended and settled particulates, and benthic invertebrates, before, during, and after this operation. Mussels, *Mytilus edulis*, were transplanted into the disposal area and monitored for trace element and chlorinated hydrocarbon content before, during, and after the experimental disposal. In addition, benthic sediments and water were also monitored for changes in chlorinated hydrocarbons during this study. Various factors affecting water quality, i.e. dissolved oxygen, pH, salinity, temperature, ammoniac nitrogen and nitrate nitrogen were also monitored before, during, and after spoiling.

The impact of disposal activities was assessed in relation to the input of pollutants from a major point source, the East Bay Municipal Utility District (EBMUD) sewage outfall. Components monitored for pollutants at the Spoil Site were also examined at this site.

At the Bodega Marine Laboratory, the uptake of the trace metals Cd, Hg, and Pb directly from seawater by three species of invertebrates was investigated. The animals were exposed to three concentrations of the above metals in water of salinity comparable to that of central San Francisco Bay. A Field-Laboratory experiment was also conducted in which animals were exposed *in situ* to water containing ambient and reduced amounts of suspended particulates. The Field-Laboratory experiment was originally designed to assess the importance of sediments and suspended particulates as pathways of pollutant uptake.

The pollutants examined during this investigation included the elements Ag, As, Cd, Cr, Cu, Fe, Hg, Mn, Ni, Pb, Se and Zn and the chlorinated hydrocarbons of the DDT and PCB groups. The primary data obtained during this study are presented in Appendix I-A as mean values with one standard deviation. Annotated and general bibliographies of references relevant to pollutant availability are included in Appendices I-B and I-C.

PART I. EXPERIMENTAL DESIGN

A. Selection and Description of Study Sites

1. Experimental Spoil Site

Upon consultation with the U.S. Army Corps of Engineers, the Regional Water Quality Control Board, the California Department of Fish and Game, and the U.S. Geological Survey, it was decided to conduct an experiment whereby sediments from an area known to be highly contaminated with various pollutants would be transferred to an area in San Francisco Bay where levels of pollutant contamination in the sediments were known to be lower. A site within the Oakland Inner Harbor (Figure 1) was selected as the source of the dredged material, since previous studies by the Corps of Engineers had shown that sediments in this area contained high concentrations of various metals. Concentrations of Cd, Hg and Pb were higher than those permitted by the U.S. Environmental Protection Agency to be present in dredge spoil material normally disposed of in San Francisco Bay. These sediments, therefore, constituted suitable material for the proposed experiment.

Since disposal of such material in San Francisco Bay might significantly increase levels of various pollutants in local food webs, a site was selected in San Francisco Bay where this material could be disposed of and where the possible effects of such disposal might be followed through time and at various distances from the disposal site. Several criteria strictly defined the choice of the experimental site. Water depths of at least 10 meters were necessary so that the site would be accessible to the dredge CHESTER HARDING. Greater depths would, also, not permit safe diving for long periods. A diverse and abundant benthic fauna was required so that possible pollutant uptake by many kinds of organisms might be followed. Fine sediments were necessary at the potential disposal site to allow efficient operation of the collecting equipment. It was crucial that the concentrations of metals and other pollutants in these sediments be lower than the concentrations recorded in the sediments of the Oakland Inner Harbor. Since the effects of dredge material disposal on pollutant availability were to be monitored, not only with time, but over an area large enough to detect diminished effects with distance from the point source of input, the area had to be large and as homogeneous as possible in all physical, chemical, and biological parameters.

A literature survey of the invertebrates, sediments, and hydrology of San Francisco Bay was undertaken to assist in the selection of potential sites. As a result of a field pre-survey (23 October - 25 October 1974) of areas chosen from the literature survey, an area between Angel Island and Berkeley, immediately east of Buoy 6

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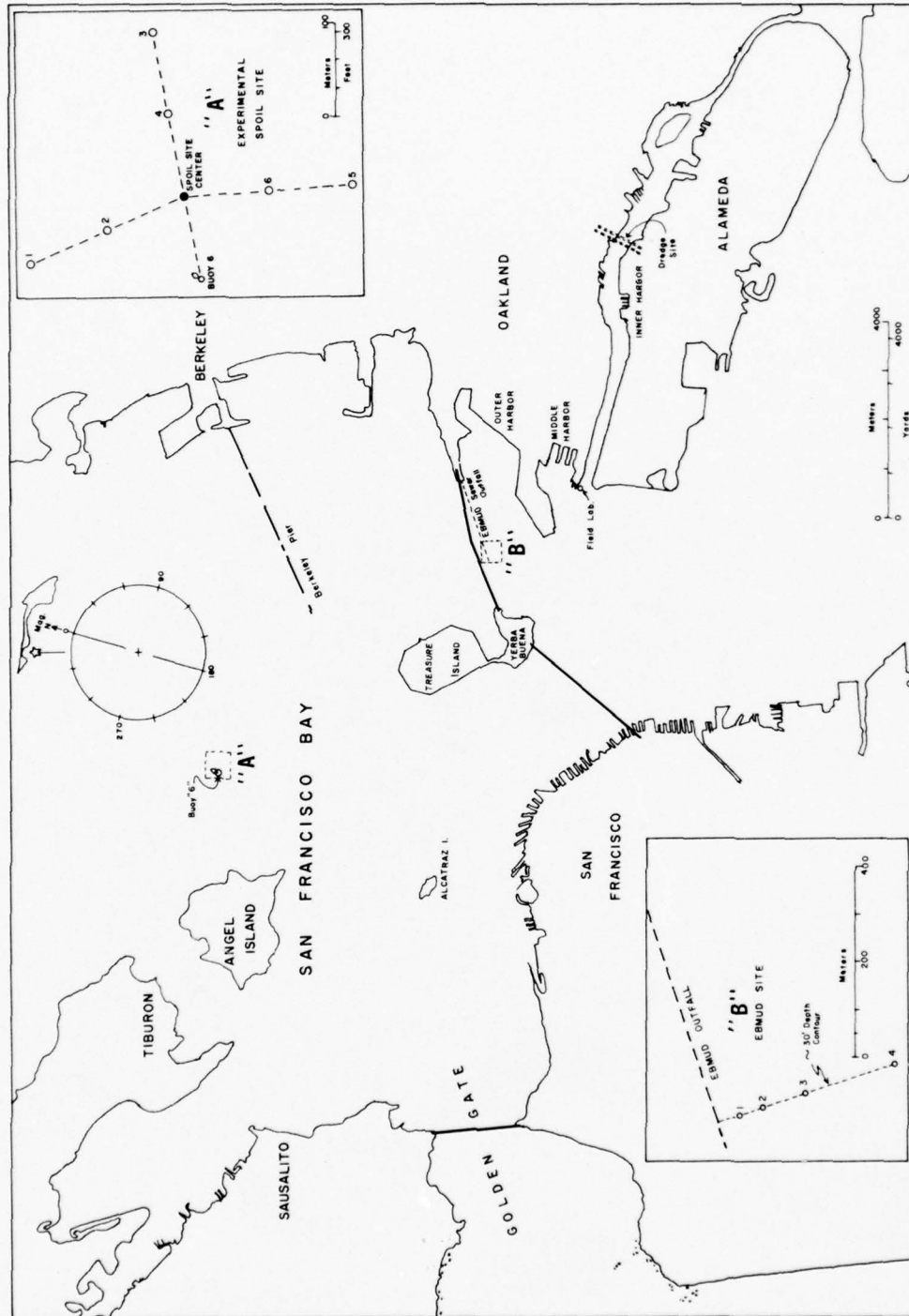


Figure 1. Map of experimental study areas in San Francisco Bay. Insets show the experimental Spoil Site area and the EBMUD site area. The Field Laboratory Site and the experimental dredge site are also indicated.

was selected as the experimental disposal site (Figure 1). This area will henceforth be called "Spoil Site".

2. East Bay Municipal Utility District Outfall

The East Bay Municipal Utility District (EBMUD) outfall is the principal waste water outlet for one of the most densely inhabited and industrialized areas of eastern San Francisco Bay. The outfall area was monitored as a means of comparing the possible influence of disposal operations on pollutant availability with the influence of the average daily discharge of 360 million liters of treated sewage on pollutant levels in local invertebrates.

A pre-survey of the EBMUD outfall area was conducted during early November, 1974. Samples were collected along the 10 meter depth contour at various distances away from the outfall to determine the abundance and distribution of benthic species.

B. Design and Establishment of Stations

1. Station Design

a. Spoil Site

The station arrangement at the Spoil Site, shown in Figure 1, Inset A, was chosen to provide the greatest spatial coverage of the area and to provide a means of comparing effects at stations immediately adjacent to the disposal impact point with those removed from this area. Six stations were established at this location; the three inner stations (2, 4, and 6) were 100 meters from a common center where disposal of dredge spoils would occur. The outer stations (1, 3, and 5) were each 300 meters away from the center; all stations were located in 10 meters of water with the exception of station 3 which was located in 7 meters of water.

The inner and outer circles of Spoil Site stations were located within the expected area of lateral dispersion of $10,000 \text{ m}^3$ of sediment to be disposed. The calculations of the expected lateral dispersion of this quantity of sediment were based on the works of Biggs (1968) and Gordon (1974). Biggs observed maximum slopes of 1:100 and average slopes of 1:500 for the dispersion of $1.1 \times 10^6 \text{ m}^3$ of spoiled sediment. According to his calculations, a 1 meter high mound of spoil would disperse horizontally between 100 and 500 m. Gordon (1974) based his calculations of lateral dispersion on observations made during seven different disposal operations which spoiled a total of $10,750 \text{ m}^3$ of sediment. He found that at least 80% of the spoil material reached the bottom within a 30 m radius when dumped in 20 m of water and that 90% of the material was within a 120 m

radius. It was assumed therefore that the disposal of 10,000 m³ of sediment of high sand content would produce a spoil mound of at least 1 m high and that the inner stations (100 meters from the impact center) would be within the resultant density current and that outer stations (200 m from disposal) would receive little or none of the spoiled material.

b. EBMUD

Four stations were established along the 10 m contour south of the EBMUD outfall (Inset B, Figure 1). Station 1 was located 50 meters south of the effluent pipe and stations 2, 3, and 4 were 100, 200, and 400 meters south of the outfall, respectively.

2. Establishment of Stations

The ten permanent stations were initially marked by small surface floats. However, owing to the heavy shipping traffic in both areas and the need for permanent platforms for the attachment of transplant animals, subsurface (2 meters below the surface) floats were constructed for all stations (Figure 2).

The subsurface floats each consisted of a styrofoam block attached by a seven meter length of stainless steel cable to two, 45 kilogram bags of "Readymix" concrete. Subsurface floats were initially located by small, inconspicuous, cork surface floats. The float locations were later marked further at the Spoil Site by white, 200 liter drums fitted with radar reflectors and flashing lights.

The subsurface floats at each site were either linked to each other or to an auxiliary float to facilitate their location. The design of these subsurface buoy systems evolved over the course of this study to overcome a variety of weather and ship traffic related difficulties. Considerable field time was spent locating stations when surface floats were lost as a result of storms or shipping traffic.

C. Field Collection Schedule

The field collection schedule shown in Table 1 illustrates the type and timing of field collections. The Spoil Site collections were designed to monitor pollutant concentrations in benthic and transplanted invertebrates and to describe the physical and chemical characteristics of waters and sediments at this site before, during, and after the experimental spoiling on 3 - 5 February, 1975.

Collections at the EBMUD outfall were scheduled as close as possible to the Spoil Site collections to facilitate intercomparison of the two areas. Transplant collections were made in conjunction

Table 1 Schedule of Field Collections.

Activity	October	November	December	January	February	March
Site Presurveys	I					
Dredge Site Sediment Collection				X		
Spoil Site					X	
Experimental spoiling						
Water collections						
Water chemistry				I	II	III
Elements and Suspended Particulates				I	II	III
Chlorinated hydrocarbons				I	II	III
Settling tube collections				I	II	III
Sediment collections			I	II	III	
Benthic invertebrate collections			I	II	III	
Mytilus edulis transplanted			X			
Transplants collected				I	II	IV
EBMUD Site						
Settling tube collections				I	II	III
Sediment collections			I		II	
Benthic invertebrate collections					II	
Mytilus edulis transplanted						
Transplants collected			X	I	II	III
Effluent collections (5 day composites)						
metals		I		II	III	IV
Effluent collection (5 day composites)				II	III	IV
chlorinated hydrocarbons			I		II	
Field Laboratory Site						
M. edulis: local and Tomales Bay						
Crassostrea gigas: Tomales Bay						
Transplanted		X				
Transplants collected		I				
Benthic invertebrate collections		I				
Sediment collections		I				
Water collections		I				
Berkeley Pier						
Mytilus edulis controls		I		II	III	IV
Collection of transplant animals			X			

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with the collection of settled particulate collectors. Benthic invertebrates and surface sediments were collected concurrently to investigate possible relationships in the pollutant levels of these components.

The data collected aboard the dredge ship CHESTER HARDING during dredging and spoiling operations are presented in Table 2. This table includes the volume of sediment disposed during each of the seven disposals and water current and tidal data for the impact center of the experimental Spoil Site at the time of each disposal.

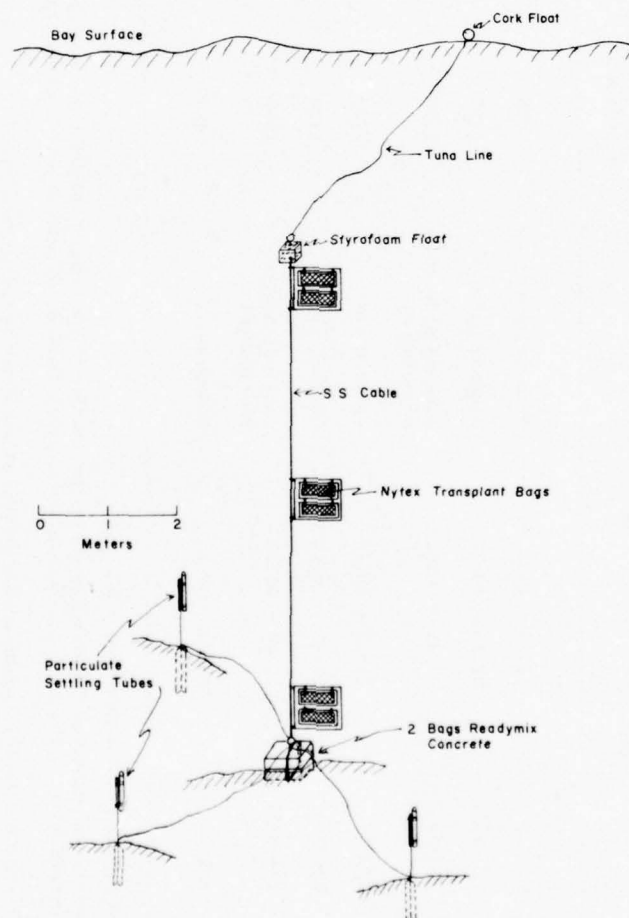


Figure 2. Diagram of one station at the Spoil Site or the 49000 site illustrating the arrangement of transplant bags and particulate settling tubes.

Table 2 Dredging and spoiling data from the dredge ship CHESTER HARDING for the experimental spoiling operation 3-5 Feb. 1975.

Load No.	Date	Collection Time a)	Collection Site b)	Cubic Meters of Sediment c)	Disposal Time	Tide	Water Velocity Knots d)
1	2/3/75	16:30 - 17:20	Alameda side	950	18:30	flood	0.8
2	2/3-4/75	22:15 - 23:32	Alameda side	1490	00:50	ebb	0.3
3	2/4/75	04:45 - 06:03	Alameda side to mid-channel	1450	07:15	flood	0.5
4	2/4/75	11:15 - 12:27	Oakland side	1450	13:50	ebb	0.8
5	2/4/75	17:15 - 18:15	Oakland side	1300	19:30	flood	0.8
6	2/4-5/75	23:00 - 00:20	Oakland side to mid-channel	1530	01:45	ebb	0.4
7	2/5/75	05:40 - 06:47	Mid-channel	1490	08:00	flood	0.5

a) All times are in Pacific Standard Time.

b) All sediment was collected immediately above the traffic tubes under Oakland Inner Harbor. The collection site is the position across the channel from which each dredge load was taken.

c) Volume of sediment in the loaded dredge hopper as measured by gauge reading.

d) Water velocity estimated from data in tide tables for the Spoil Site area.

PART II. EXPERIMENTAL METHODS

A. Field Collection Methods

1. Collection of Benthic Invertebrates

The large number of benthic organisms necessary for monitoring of trace element concentrations were collected by a specially designed diver-operated suction dredge. The device, similar to those used in underwater gold mining, was constructed entirely of PVC (polyvinyl-chloride) plastic pipe, and all dredge parts and fittings were constructed either of PVC, Teflon or Nylon to eliminate possible metal contamination.

The 16 cm diameter by 100 cm long dredge was powered by a 7,800 liter/min. centrifugal water pump which forced water through the dredge and created the necessary suction. A 1.5 meter long, 1.0 mm mesh Nytex bag was fitted to the discharge end of the dredge which retained benthic organisms and some sediment. Samples collected by this device were brought to the surface and washed on a 2.0 mm mesh corrosion-free, stainless steel screen. The invertebrates retained on this screen were sorted and returned alive to the Lawrence Berkeley Laboratory.

2. Collection of Benthic Sediments

Benthic sediments were collected as indicated in the collection schedule presented in Table 1. Three replicate sediment samples were obtained from each station during each sampling period for the determination of particle size frequency and heavy metal content and for the determination of chlorinated hydrocarbons.

Sediment core samples for particle size and metal analyses were obtained by divers using 60 or 30 cm long by 10 cm I.D., PVC pipe. These cores were fitted with PVC caps having a 1 cm hole in the center for rubber stoppers. Divers inserted the unstoppered cores vertically into the sediment and when the top of the device was flush with the sediment surface, the stoppers were inserted and the cores carefully removed. The cores were then capped, held vertically, and returned to the boat.

On board, each sediment core was extruded into a clean plastic tray by an acrylic plastic plunger inserted from the bottom. Each sediment core from the 60 cm tubes was divided into four vertical layers; 0-2 cm, 2-10 cm, 10-30 cm and 30-60 cm. The 0-2 cm layer was transferred with a plastic scoop into a large size Whirlpak (polyethylene) bag. Each of the other three layers was first thoroughly mixed and then a 150 ml subsample of each respective layer was trans-

ferred to a Whirlpak bag. After the initial collections at the Spoil Site and EBMUD Stations, only the 30 cm long cores were used and only the 0-2 cm layers were retained for metals and particle size analyses.

Sediment samples for chlorinated hydrocarbon analyses were collected with a 15 cm length of precleaned, 5 cm I.D., aluminum tubing. A large size, Whirlpak bag, previously shown to be free of organochlorine contamination) was held open over one end of the tube by the diver as he scraped the other end of the tube horizontally along the surface of the sediment until the bag was filled. Penetration of the tube into the sediment was maintained at between 2-3 cm.

Sediment samples were collected from the Oakland Inner Harbor dredge site prior to the dredging operation to determine the particle size distribution, the trace element content, and the chlorinated hydrocarbon concentrations in sediments to be dumped at the experimental spoil site. It was originally planned to collect 18 vertical cores of at least 1.5 m in length, with a 5 cm I.D. aluminum tube, from within the designated dredge zone (Figure 1). However, hardpacked clay was encountered at a depth of 10-20 cm below the surface of the sediment which permitted only core lengths of less than 20 cm. These cores were sealed with polyethylene caps and returned to the laboratory where the collected sediments were extruded and composited into three replicate samples of six cores each. Each of these samples was thoroughly mixed and divided into two equal fractions; one for trace element and particle size distribution analyses and one for chlorinated hydrocarbon determinations.

3. Collection of Settled Particulates

Settled particulates were collected at the Spoil Site and EBMUD stations by settled particulate traps similar in design to those developed and tested by Mr. Anthony Weaver (unpublished manuscript) at the Hopkins Marine Station.

The settled particulate traps employed at Spoil Site and EBMUD stations were constructed of 50 cm long by 5 cm I.D. lengths of acrylic tubing, closed on the bottom by a polyethylene cap and open at the top. The tube length of ten times the diameter provided sufficient dead space above the collected particulates to prevent resuspension and loss from scouring. A 40 cm length of 1.27 cm I.D. flexible PVC tubing was taped to one side of the sediment traps and stoppered on top. The tubing could be slipped over a 1 cm diameter reinforcing rod (rebar) and served as the mounting for these traps. Holders for the particulate traps were constructed of angle iron and rebar. Sections of 6.4 x 6.4 x 0.5 cm angle iron 60 cm long were welded to one end of 1.5 m lengths of the rebar and painted with epoxy paint. In the field, each section of angle iron was forced into the sediments, and the protruding rebar was then vertically aligned. A line was tied between the holder

and the subsurface buoy anchor to facilitate recovery of the traps. The traps were positioned so that the open ends were 1 meter above the sediment surface.

Three sediment traps were installed at each of the ten stations at distances of 2-3 meters from the subsurface buoy anchors (Figure 2). Recovery of the sediment traps closely corresponded in time with the collection of transplanted M. edulis (Table 1).

On recovery, the traps were capped by a diver, removed from the rebar holders, and replaced by clean traps. The traps were then returned to the laboratory where the settled particulates and overlying water were transferred to clean one-liter glass jars fitted with polyethylene liners and plastic lids. These samples were stored at 4°C for later analysis.

4. Collection of Water and Suspended Particulates

The concentrations of metals in solution (including metals on particles 0.07μ) and elements associated with suspended particulates of three size classes: coarse ($>4\mu$), medium (4μ to $\geq 1\mu$), and fine (1μ to $\geq 0.07\mu$) were measured at the Spoil Site before, during, and after the experimental disposal. This information was necessary to assess the potential availability of trace elements in the solute phase and those associated with the various size classes of particulates in dredged material. Water samples including suspended particulates were also collected for determinations of chlorinated hydrocarbons.

a. Collection of water for trace element analyses

A specially constructed pumping system was used to collect water and suspended particulates. This system consisted of a high volume peristaltic pump, a 15 m length of thick-walled 0.8 cm I.D., PVC flexible tubing, and a weighted collecting device ("fish") which held the intake end of the collection tube. The "fish" consisted of a 1 m length of 1.3 cm I.D., PVC pipe with four large PVC fins welded to the one end. One end of the intake tubing extended 5 cm beyond the front end of the "fish". The intake tube was covered with plastic bags while being lowered through the water surface to avoid possible contamination. The "fish" was towed with a plastic-coated stainless steel cable weighed with a 15 kg epoxy-coated lead ball suspended two meters below the "fish". In operation, the "fish" was balanced to remain horizontal and faced the prevailing currents owing to the stabilizing fins and a swivel mounting. The amount and angle of cable lowered were used to estimate sampling depths to the nearest meter.

Assuming a specific gravity of suspended particulates of 2.65, an intake water velocity of 1.2 m/sec. is sufficient to insure an

unbiased collection of particulates up to 80 μ in diameter (Shelley and Kirkpatrick, 1973). Water samples for the determination of suspended particulates were collected in 25 liter, acid-leached polyethylene carboys. Water samples to be analyzed for metals in 'solution' were collected in 250 ml, acid-leached FEP Teflon bottles (Walgene Corp.).

In the pre-spoiling collection, samples were collected over a 24-hour period at 7 meters and at 3 meters above the bottom at each of the Spoil Site stations. At each station and depth, 6 liters of water were collected every 6 hours and composited into one 25 liter carboy to obtain an estimate of the average particulate load and element concentrations over a 24-hour period.

During the experimental disposal, particulate samples from two depths at all 6 stations were collected as above, but the compositing method differed as follows: at forty-five minutes and again at four hours after each of the seven experimental disposals, a 1.8 liter water sample was collected at two depths at each of the six stations. At station 1, 250 ml samples were collected at two depths, 4 hours after the seventh disposal for the determination of metals in solution.

Five water samples were collected 4-5 m above the bottom at the center of the Spoil Site during the disposal operation. One 24 liter sample (T_{0.08}) was collected five minutes prior to the first disposal. Samples T₀, T_{1.5}, T₃, and T_{4.5} were composites of 3.5 liter sub-samples collected 5 minutes after, 1.5 hours after, 3 hours after, and 4.5 hours after each of the seven disposals. Sample T₀ was collected from within the disposal plume. A sixth sample (T₉) consisted of one, 48 liter sample collected 9 hours after the last disposal. These samples provided data on the magnitude and residence time of the particulates associated with disposal operations.

Except for Cu, the element concentrations in suspended particulates of top samples were not significantly different from bottom samples during the pre-spoil collection. Therefore, only bottom samples were taken during the post-spoil collection. Owing to the uniformity of load and dry weight concentrations of metals observed between stations in pre-spoil samples, particulate samples from the three inner stations were composited into a single sample during the post-spoil collection. Outer station collections were treated similarly. At each of the inner and outer stations, 4.2 liters of water were collected every 6 hours and composited into an inner circle sample and an outer circle sample. The composited volume of each of these samples totaled 50 liters.

Two 0.5 liter composite water samples were collected for metals in "solution"; one from inner stations and one from outer stations. The depth and timing of these composites were similar to post-spoil suspended particulate composites. All samples were stored in the laboratory at 4°C prior to preparation and analysis.

b. Collection of water for chlorinated hydrocarbons

The water samples collected for chlorinated hydrocarbon analysis included suspended particulates which were not separated before analyses. Chlorinated hydrocarbons were extracted in situ by pumping water through a stainless steel column 30 cm long x 6 cm I.D. containing 5 plugs, 6 cm diameter x 6 cm long, of high density polyurethane foam with an 81% ether content (United Foam Company, Los Angeles). Foam plugs were pre-cleaned with acetone by Soxhlet-extraction, packed in columns and rinsed with hexane until blanks of less than 0.5 picograms of PCB per sample were obtained. Techniques of mounting the columns, pumping, and other shipboard procedures are described by de Lappe and Risebrough (unpublished manuscript). The pumping rate was 650 ml/minute and 40 liter samples were collected. The column was lowered 5 to 8 meters below the surface during the pumping period to obtain representative samples of the water column. After collection, the columns were wrapped with aluminum foil until analysis.

One to two hours of pumping was required per sample. Pre-spoil collections were alternately made at inside and outside Spoil Site stations over a 24-hour period. During the spoiling operation, one sample was collected at the center of the disposal area immediately following each of the seven disposal operations. Water samples were collected at stations down current and up current from the impact center between disposals. Post-spoil water samples were also collected at Spoil Site stations over a 24-hour period.

B. Preparation and Analysis of Samples

1. Preparation of Invertebrates for Trace Element Analyses

The live invertebrates were immediately returned to the laboratory and purged in clean, plastic trays fitted with Nylon screens. The screens elevated the organisms and separated them slightly from their fecal material to discourage coprophagy.

The animals (except Ampelisca milleri and Stylatula elongata which could not live under these conditions) were purged of ingested sediment for three days in 12°C water collected from the same area as the animals. Purging was necessary to rid the animals of sediments contained in their stomachs and intestines so that only elements associated with tissues would be analyzed. The animals were separated by species and station. The purge water was changed daily and continuously aerated. The length of time necessary for purging the majority of sediments from Macoma nasuta and Pectinaria californiensis, and the rate of trace element desorption from tissues in M. nasuta were determined in experiments described in Part II, Section E.

Pelecypods were shucked and placed in clean plastic vials for

freeze-drying and their shells were saved for size measurements. Polychaetes and other small invertebrates were also freeze-dried in plastic containers.

Analyses of individual animals by Anderlini *et al.* (1975) and others have revealed considerable variance in the concentrations of various metals among individuals within a species. Therefore, when possible, at least 10 individuals of a species were composited into each of three replicate samples for each station and collection. This method was designed to increase the analytical precision of the determinations and to assure as normal a distribution of data as possible. This technique was an application of the Central Limit theorem and is discussed in Part II, Section D.

The *Mytilus edulis* transplants were also returned alive to the laboratory for purging and preparation for metals analyses. The mussels from each station and depth were placed in clean plastic trays, and purged for three days as described above. After purging, the mussels were shucked and separated into three replicates of 10 animals each.

A digestion step was necessary to break down the tissues of clams, worms and mussels before freeze-drying. Approximately 3-5 ml (dependent on sample size) of the strong organic base, 0.012 M TMAH (tetramethyl-ammonium hydroxide, Eastman-Kodak) were added to each sample. The samples were then heated at 40°C for 8 hours, after which they were freeze-dried for 3-4 days.

The freeze-dried samples were then ground to a fine powder in their containers with a stainless steel pestle. The ground material was sifted through a 0.5 mm mesh Nylon screen to remove unground connective tissue and muscle fibers and to increase sample homogeneity. At this point, the samples were analyzed for Hg by the IZAA system described in Part II, Section C. Samples for the other elements were prepared by pressing 150 mg of the finely ground powder into a thin, 25.4 mm diameter wafer, using a stainless steel die and a 10,000 p.s.i. hydraulic press. The wafers were then analyzed on the X-ray fluorescence system described in Part II, Section C.

2. Preparation of *M. edulis* for Chlorinated Hydrocarbon Analysis

The mussels which were analyzed for chlorinated hydrocarbon concentrations were separated from mussels used for element determinations and immediately frozen upon return to the laboratory. Each sample consisted of three replicates of 10 animals each. The tissues of the 10 mussels were pooled in a small beaker and dissolved over steam with BFM solution (Stanley and Le Favoure, 1965). After dilution with an equivalent volume of water, the extract was transferred to a 125 ml separatory funnel and extracted with three, 50 ml volumes of 15% ethyl ether in hexane. The combined organic phases were reduced to

approximately 10 ml and transferred to a glass centrifuge tube containing 2 ml of concentrated sulfuric acid. After shaking, the samples were frozen in a dry ice/acetone slurry and the organic phase was decanted from the sulfuric acid into a graduated cylinder for gas chromatographic analysis.

3. Sediments and Settled Particulates

a. Size distribution analysis

The benthic sediment samples collected for particle size distribution and trace element analyses were returned to the laboratory and drained of excess water. After thoroughly mixing each of these samples, aliquots were removed and immediately frozen for trace element determinations.

The remainder of each of the wet sediment samples was used for the determination of particle size distribution using standard procedures (Galehouse, 1971; and Ingram, 1971). The samples were initially treated with 35% hydrogen peroxide to digest organic matter and washed in dialysis bags to remove soluble salts. The fines were separated from sand by wet sieving through a 0.062 mm mesh screen, treated with 10% sodium hexa-meta-phosphate, and dispersed in distilled water. The samples were kept in a constant temperature water bath and the percent weight of silt, coarse clay and fine clay in these dispersed sediments was then determined by pipette analysis.

Samples of settled particulates were stored in the laboratory at 4°C to retard biological activity until analysis. Each sample was suspended in a clean graduated cylinder with twice distilled water and the volume was made up to 1000 ml. The mouth of the cylinder was covered with a plastic sheet and the sample was mixed by repeated inversions for two minutes.

Five seconds after mixing, a 50 ml aliquot was drawn from a depth of 20 cm with a volumetric pipette. The mixing procedure was repeated and another 50 ml aliquot was obtained. These respective aliquots were centrifuged and the residues were washed three times with distilled water to remove retained salts. The aliquots were transferred to evaporating dishes and dried to a constant weight. Deposition rates in Mg settled particulates/cm²/day, were estimated from these aliquots. The remaining particulates were removed from the solution by centrifugation and prepared for metals analyses.

b. Analysis of sediments and settled particulates for trace elements

A 15 ml subsample was removed from each sediment and settled

particulate sample, transferred to a clean, plastic, snap-top vial and frozen for trace element analysis. These subsamples were freeze-dried and a 75 mg aliquot was weighed out and mixed with 75 mg of pure cellulose in an acid-cleaned, agate mortar and pestle. The two materials were ground together and pressed into a wafer for X-ray fluorescence analysis for the elements: Ag, As, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Se, and Zn. The remainder of the freeze-dried sediment was analyzed for Hg on the IZAA system.

4. Preparation of Water and Suspended Particulates for Trace Element Analyses

a. Suspended particulates

Particulates were separated from water samples by centrifugation. Centrifugation parameters for batch separation were rotor speed and time, while for continuous flow operation, the parameters were rotor speed and flow rate. Estimation of values of these parameters for spherical, anhydrous particles took into account the effects of buoyant forces and viscous drag forces as given by Stoke's resistance law (Berman, 1966). Particle size estimates were also based on formulae presented by Berman (op. cit.). As is the case of all analyses of particle size based upon settling rates, the size or equivalent spherical diameter (esd) of suspended particulates is an operational definition.

Standard methods used to prepare sediment samples for size analysis include deflocculation, organic digestion, and vigorous stirring (Galehouse, 1971). With the exception of the organic digestion step, these procedures were followed in the preparation of water samples for size analysis of the suspended particulates.

Individual, 24-liter samples were equilibrated to 20°C and deflocculated by the addition of 400 ml of 0.4N(NaPO₃)₆. During the first hour after deflocculation, the bottles were shaken several times to ensure mixing. After standing for 24 hours, the samples were stirred for one-half hour with an electro-polished, nitric acid-rinsed, stainless steel paddle device. The paddle protruded through a .48 cm diameter hole in a special carboy lid and was driven at 800 RPM.

Immediately following stirring, a Beckman J21B centrifuge equipped with a Beckman JCF-Z continuous flow, titanium rotor was used to separate suspended particulates into three size classes. Each sample was passed three times through the rotor; each pass removed one distinct size class. First pass: particles $\geq 4\mu$ esd were removed; flow rate (Q) was 740 ± 10 ml/min. and the rotor speed was 500 ± 15 RPM. Second Pass: particles between 4μ and 1μ esd were removed; Q = 740 ± 10 ml/min. and the rotor speed was 1970 ± 20 RPM. Third pass: particles between 1μ and 0.07μ esd were removed; Q = 400 ± 10 ml/min.

and the rotor speed was $19,000 \pm 300$ RPM.

A Master-Flex, variable flow peristaltic pump was used to control the centrifuge flow to within 10 ml/min. Polyethylene lines were used, except in the pump body where a short section of food grade Tygon tubing was necessary. The flow was continuously monitored with a Gilmont flow meter; sample volumes were determined by weighing. The rotor and sample temperatures were maintained between 21° and 23°C during centrifugation. For the first and second passes the RPM were set and checked with a General Electric strobe light. All particle size determinations were based on an assumed specific gravity of 2.65. For organic particles with a specific gravity of 1.07, the equivalent size classes would be: 223μ , 23μ to 5.7μ to 0.4μ .

The sediment removed in each pass was washed and concentrated into a small volume by repeated batch centrifugation, decantation of saline supernatant, washing with distilled water and re-centrifugation. This washing process was repeated twice or until the chloride ion concentration was between 10 ppm and 1 ppm as determined by a visual chloride ion precipitation test with AgNO_3 . Further reduction of the chloride concentration in the supernatant³ caused the sediment to peptize, making recovery difficult.

Removal of salts from the centrifuged sediment and interstitial water was essential for the accurate determination of dry weights of the particulates in each size class. The 1 to 10 ppm chloride concentration in water remaining with sediment, after decanting on the final wash step, was estimated to contribute less than 0.2% to the final dry weight of the particulates. Tests were conducted on three suspended particulate samples to determine whether particulate metal concentrations were affected by deflocculation and/or removal of salts. Subsamples of each of these samples were centrifuged with and without prior deflocculation and particulates in the three size ranges were analyzed by X-ray fluorescence. This was repeated on subsamples with and without salt removal. The differences in particulate metal concentrations for subsamples receiving different treatments were, in all cases, less than or equal to the counting errors associated with the X-ray fluorescence analysis of individual metals.

Samples were dried for 48 hours at 110°C , equilibrated to room temperature in a dessicator and weighed. Thirty to fifty mg of each of these samples were mixed with a known amount of cellulose and made into wafers for X-ray fluorescence analyses. Mercury was determined by IZAA analysis on separate portions of these samples.

5. Water

a. Determination of Cu, Cd and Pb in Water

The 250 ml water samples collected for the analysis of metals

in solution were immediately returned to the laboratory, equilibrated to 20°C, and transferred to acid-washed, 50 ml polyethylene centrifuge tubes. These samples were centrifuged for 1.8 hours at 7500 RPM and 20°C using a Beckman J21B centrifuge and a J7.5 batch rotor. Particles with esd's $>0.07\mu$ ($p = 2.65$) and $\geq 0.4\mu$ ($p = 1.07$) were removed from the solutions. The top 30 ml of supernatant was transferred to a clean, acid-leached Teflon bottle and acidified to a pH of 2 with 10 N, 3 x distilled HCl (National Bureau of Standards). Samples for Cd, Cu, and Pb determinations were heated in a water bath for three days at 55°C and then stored at 4°C until analysis. Samples for mercury analysis were also stored at 4°C until analysis.

Cadmium, Cu, and Pb were extracted from centrifuged and acidified water samples using the APDC/MIBK ammonium pyrrolidine dithiocarbamate (also designated as pyrrolidine carbodithioate) / methyl isobutyl ketone method of Brooks, *et al.* (1967) with the nitric acid back-extraction method described by Segar and Gonzales (1972) and Serne and Mercer (1975). Lead and Cu concentrations in back-extracted acid samples were determined on a Perkin Elmer Model 303 Atomic Absorption Spectrophotometer equipped with a heated graphite atomizer (Model HGA 2000) and deuterium background corrector. Cadmium and Hg determinations were made on Isotope-Shift Zeeman Atomic Absorption Spectrophotometers.

The details of the extraction procedure for Cd, Cu and Pb are as follows: 100 ml of sample (pH 2.1 to 2.3) were transferred to an acid-leached, Pyrex separatory funnel equipped with a Teflon stopcock, and buffered to pH 4.4-4.8 with 0.5 ml of 5 M, purified sodium acetate. One ml of 1% APDC, prepared and purified daily, was added and the funnel shaken for 30 minutes on a Burrell wrist shaker after which the phases were allowed to separate for 20 minutes. The lower aqueous phase was removed, the pH measured (samples remained between pH 4.4-5.8) and subsequently stored in an acid-leached Teflon bottle for use in preparation of standards (see below). To the remaining MIBK phase, 3 ml of 6 M NBS 2 x distilled HNO_3 were added and the funnel shaken for an additional 20 minutes. The phases were allowed to separate for 15 minutes and the acid phase transferred to acid-leached Teflon bottles and stored in the dark at 4°C until analysis. The MIBK phase was discarded. The acid back-extraction step was used since chelated metals are unstable in the MIBK phase. The acid back-extracted samples remained stable for up to two weeks.

Standards were prepared from the aqueous phase of extracted samples. This once extracted water was extracted a second time with 1 ml of 1% APDC and 5 ml of MIBK and back extracted with 3 ml of 6 M HNO_3 , as described above, to remove any remaining metals.

These twice-extracted water samples were composited and 100 ml portions were spiked with increments of stock standard solutions. These standard San Francisco Bay water samples were extracted as above.

The amount of MIBK was reduced from 7 ml to 5 ml since the aqueous phase was already saturated with MIBK at room temperature. The back-extracted water standards were stored as above.

Since our extraction procedure was identical, except for the use of acetate buffer, to that used by Serne and Mercer (1975) on San Francisco Bay water, we have assumed that the extraction efficiencies of various metals were similar to theirs. Successive extractions on water samples showed that the average percent extraction of stable Cu, Cd, and Pb for the APDC/MIBK/HNO₃ system were 90%, 90%, and 100% (1st extraction); 9%, 8%, and 0% (2nd extraction) and 1%, 2%, and 0% (3rd extraction) (Serne and Mercer, 1975).

Two successive APDC/MIBK extractions were performed on the aqueous phase of several of the Spoil Site samples. The MIBK phase from the second extraction was back-extracted and analyzed for Pb, Cu, and Cd. No absorbance signal (<0.004 absorbance units) was observed.

On the basis of these experiments, one extraction was sufficient for samples and two extractions sufficient for the preparation of water to be used for standards. Since water standards were extracted by the same procedure as the original samples, the method was self-compensating in that it took into account the incomplete extraction of original sample and standards alike.

Purified sodium acetate crystals were prepared by recrystallization of a saturated solution of sodium acetate. The 1% solution of APDC was prepared and purified daily using the method of Smith and Windom (1967). Deionized 2 x quartz-distilled water (Q-water) was used to prepare the APDC and saturated sodium acetate solutions. All Pyrex and Teflon labware was leached twice in concentrated HNO₃ at 75°C, rinsed with Q-water, leached at 55°C in 1% 2 x distilled NBS HNO₃, rinsed and stored in plastic wrappers.

Polyethylene containers were washed as above, leached in concentrated HCl at 20°C, rinsed with Q-water, leached in 1% G. Frederick Smith 1 x distilled HCl, rinsed, leached in 0.1%, 2 x distilled NBS HCl, rinsed and stored as above. Leaching times in all baths were 4 days or longer and stored bottles were filled with Q-water until used.

The concentrations of Pb and Cu in acid back-extracted samples were measured in 10 µl samples injected into the Perkin Elmer HGA 2000 graphite furnace. For Pb, the optimum settings for drying, charring, and atomization were found to be 125°C for 60 seconds, 500°C for 20 seconds, and 2100°C for 10 seconds, respectively. For copper, the optimum settings were: drying at 150°C for 60 seconds, charring at 600°C for 20 seconds, and atomization at 2325°C for 10 seconds.

Lead, Cu, and Cd reagent blanks were determined by acidifying, buffering, and extracting Q-water samples. No absorbance signal (<0.004 absorbance units) was observed. To evaluate possible Pb, Cu, and Cd contamination from acid-leached sample bottles and the centrifugation procedure, a second set of Q-water samples was

centrifuged, put into Teflon sample bottles, acidified to a pH of 2, heated at 55°C for 3 days and allowed to stand for 2 months. These samples were then buffered and extracted as above. Again, no absorbance signal was observed. The detection limit for Pb, Cu, and Cd was 0.1 µg/l, owing to a lack of precision in low concentration standards and the subsequent error in determining the slope of the standard working curves.

b. Determination of mercury in water

Mercury was determined in water samples by a modification of the EPA (Anonymous, 1971) method for the determination of Hg in solution (Hatch and Ott, 1968). This method was adapted for use with the Isotope-Shift Zeeman Atomic Absorption (IZAA) apparatus described in Part II, Section C.

Since the absorption tube configuration in the IZAA instrument did not lend itself to circulation of gases as required by the Hatch-Ott procedure, mercury vapor was sparged by nitrogen or argon from the reduced sample through a stainless steel collecting tube containing approximately 150 mg of finely chopped gold foil or crumpled 1 mil gold wire. Before each use, collecting tubes were heated at 800°C until satisfactory blanks containing no detectable mercury were obtained. The gold was tamped in place with a clean glass rod after each heating cycle. With an argon flow rate of 500 to 1000 ml/min., the mercury recovery was usually in excess of 90%; the use of two collectors in series increased the overall collection efficiency to near 100%.

A description of the method employed is summarized below:

1. A 40 ml water sample was transferred to a clean 32 mm x 200 mm test tube. If the expected Hg concentration was greater than 0.5 ppb, a smaller sample was used and diluted to 40 ml with 2X Q-water.
2. Two ml of conc. H_2SO_4 and 1 ml of conc. HNO_3 were added to each 40 ml sample.
3. Next, 0.4 ml of KMnO_4 solution (5% w/v) was added and the solution allowed to stand for 15 minutes.
4. 0.8 ml of $\text{K}_2\text{S}_2\text{O}_8$ solution (5% w/v) was added and the solution heated for 20 minutes at 60°C and allowed to cool for 10 minutes.
5. Next, 0.8 ml of $\text{H}_2\text{NOH-HCl-NaCl}$ (12% each w/v) was added to each sample. The solution was stable at this point and could stand for several hours before proceeding to the next step.
6. The pre-purged collectors, aeration device, etc. were arranged as in Figure 3. Two ml of SnCl_2

reagent were added to the sample and the aeration tube and stopper immediately inserted. The sample was sparged for 5 minutes with 500 ml/min. Ar or N₂ gas.

7. The collecting tubes were removed and placed in sealed plastic holders for immediate analysis on IZAA.

Calibration curves of total mercury chloride recovered vs. mercury chloride added to quartz distilled water and San Francisco Bay water respectively are shown in Figure 4. The recovery is excellent. Surprisingly, the results show less mercury in San Francisco Bay water than is present in our laboratory conductivity water. A reagent blank was determined by addition of 1, 2 and 3 times the normal amount of reagents and it was determined that the bay water contained 10 parts per trillion mercury (10 ng/liter). This result agrees with recent determinations of mercury in Puget Sound water by Bothner and Robertson (1975).

c. Determination of water quality parameters

Water collections for dissolved oxygen, pH, nitrates, and ammonia determinations and in situ measurements of temperature and salinity were made hourly over 24-hour periods on 8 - 9 January, 1975 and on 3 - 4 March, 1975. During the experimental disposal, hourly samples were collected during the entire 42-hour period on 3-5 February, 1975. Since physical and chemical conditions may affect the rates of uptake and depuration of pollutants, these measurements were considered important for understanding any changes in the pollutant levels in the benthic and transplanted animals monitored during this study.

During these sampling periods, the vessel moved alternately from inner to outer stations where in situ measurements were made at one meter from the bottom and at one meter below the surface. Additional measurements and collections were made within the plume immediately following each of the seven experimental disposals to obtain data on the maximum effects of dredge disposal operations on the above parameters. Temperature and salinity were determined with a Yellow Springs Model 33Y temperature-salinity probe. Water samples were collected in Van Dorn water samplers. Salinity measurements were checked against argentimetric salinity titrations (Strickland and Parsons, 1972, pp. 1719) in the laboratory. Salinity samples were frozen in glass jars until titration.

Subsamples for nitrate determination were stored in glass jars on ice in the dark until returning to the laboratory. These samples were analyzed by the Brucine-Sulfate Method within 16 hours of collection (Tanas et al., 1971, pp. 461-464; and Jenkins and Mesker, 1964, p. 610).

The Van Dorn bottles were also used to collect water for the

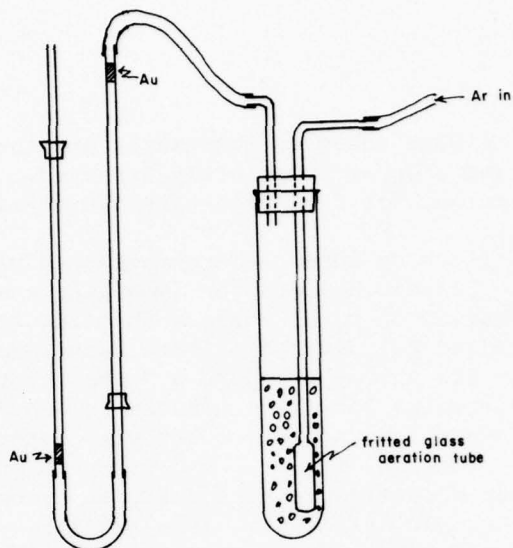


Figure 3. Arrangement for sparging mercury from reduced water samples and collecting the mercury on gold mesh contained in 3/16 inch diameter thin wall stainless steel collector tubes. Connections are with Tygon tubing.

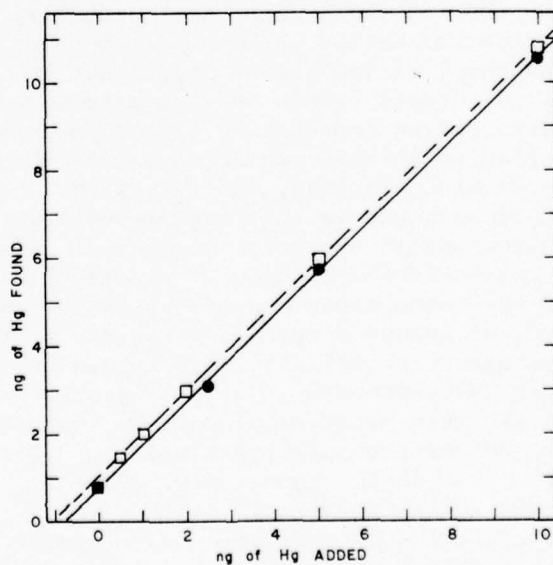


Figure 4. Recovery of mercury added to 40 ml samples of conductivity water (squares) and San Francisco Bay water from the Spoil Site (filled circles).

determination of nitrogen as ammonia. Subsamples were stored in plastic vials and analyzed by a method slightly revised from the phenol-hypochlorite method described by Strickland and Parsons, (1972, pp. 87-89). The thorough rinsing of the measuring cylinder with distilled water described in this method was preceded by an initial rinse with 10% HCl to prevent NH_3 contamination.

Additional water samples collected in Van Dorn samplers were carefully transferred to BOD (biological oxygen demand) bottles and immediately fixed with manganous sulphate and alkaline iodide solutions for later dissolved oxygen determinations. Determinations of dissolved oxygen were performed by a standard Winkler titration (Strickland and Parsons, 1972, pp. 21-26). All dissolved oxygen determinations were made within 16 hours of collection.

C. Analytical Methods

The analyses of trace elements in all samples of organisms, sediments, and suspended and settled particulates were performed on the X-ray fluorescence and Isotope-Shift Zeeman Atomic Absorption systems at Lawrence Berkeley Laboratory. These instruments offer many advantages for the rapid determination of a suite of elements in a large number of samples. Both systems require relatively little sample preparation and low sample weight. The X-ray fluorescence system allows the simultaneous analyses of several elements.

1. X-ray Fluorescence Spectrometry (XRF)

A semiconductor detector, X-ray fluorescence spectrometer was used for the determination of the elements Ag, As, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Se, and Zn in this study. This system employed a low power, tungsten (W)-anode, X-ray tube (30 watts) which was operated at 80 KV. Interchangeable secondary targets were utilized to obtain near, monochromatic excitation radiation which enhance sensitivities over those obtainable by other X-ray systems.

Determination of the elements As, Cr, Cu, Fe, Mn, Ni, Pb, Se and Zn were made with a Mo (molybdenum) secondary target. A Tb (terbium) secondary target was used for the Ag and Cd analyses. Transmission measurements were made on representative samples of each invertebrate species and sediment type to determine the corrections necessary to compensate for matrix absorption effects (Giauque *et al.*, 1973).

Data obtained on this system were recorded on magnetic tape and subtraction of scattered excitation radiation background, unfolding of X-ray line overlap, and concentration calculations with counting errors, were carried out with the Lawrence Berkeley Laboratory's CDC 7600 computer. A more detailed description of the theory and operation of this semiconductor detector X-ray system is presented by Jaklevic

et al. (1973).

Below is a list of counting errors (one standard deviation) and detection limits (95% confidence limits based on counting errors) for various elements in biological samples as calculated from specimens collected and analyzed during this study. The counting errors given are averaged from analyses of M. edulis, P. californiensis, M. nasuta, A. milleri and S. elongata.

Element	Counting Error	Detection Limit (PPM)
	(PPM)	2X counting error
Ag	0.6	1.2
As	0.7	1.4
Cd	0.6	1.2
Cr	3.6	7.2
Cu	1.3	2.6
Fe	7.0	14.0
Mn	3.1	6.2
Ni	1.1	2.2
Pb	1.4	2.8
Se	0.6	1.2
Zn	2.3	4.6

Reproducibility of X-ray data was checked by two methods. A standard wafer, made from lyophilized tissue of M. edulis, was analyzed over a number of different days to determine whether humidity fluctuations affected the geometry of the sample wafers. Changes in wafer weight and shape due to humidity fluctuations are known to influence X-ray sensitivity (R. Giaque, personal communication). Duplicate wafers of single samples were analyzed to determine possible variation due to sample preparation or sample inhomogeneity. The results of these analyses are presented in Tables 3 and 4.

2. Isotope-Shift Zeeman Atomic Absorption (IZAA)

Mercury determinations in all biological, sediment, and water samples were performed on a new type of atomic absorption spectrometer developed by Dr. T. Hadeishi at Lawrence Berkeley Laboratory. This instrument allowed the measurement of the Hg content of a solid or liquid sample without previous chemical separation of the Hg from the host material (Hadeishi and McLaughlin, 1971).

In this technique, the sample is thermally decomposed in a nickel furnace maintained at a temperature of 800°C. The decomposition products are swept into a heated absorption tube by oxygen carrier gas, where they are scanned by a light beam from a ^{204}Hg electrodeless-discharge lamp operated in a magnetic field. The light emitted from this lamp has three components; one has a wave length centered on the

Table 3 Analytical methods. X-ray fluorescence spectrometry. Variability of trace element concentrations determined by x-ray fluorescence spectrometry associated with multiple analyses of a single wafer from a *Mytilus edulis* sample. Analyses were made over a period of 2 months. Values in ppm dry weight are presented.

Element	Number of Analyses	Mean Concentration (ppm)	Mean Counting Error (ppm)	Standard Deviation (ppm)
Ag	25	1.6	0.4	0.6
As	19	7.8	0.6	1.2
Cd	25	5.2	0.5	0.7
Cu	19	8.2	0.9	0.7
Fe	19	145	3.3	6
Mn	19	5.9	2.3	2.0
Ni	19	2.3	0.9	0.6
Pb	19	3.4	1.2	1.9
Se	19	9.9	0.6	0.6
Zn	19	203	2.1	7

Table 4 Analytical Methods. X-ray fluorescence spectrometry. Results of metals analyses of three replicates from one sample of *Macoma nasuta* tissue analyzed during two periods. Mean concentrations in ppm dry weight and one standard deviation () are presented. Mean counting errors are also included.

Sample Identification		Number of Determinations	Ag	Cd	Cu	Fe	Mn	Pb	Zn
<u>Macoma nasuta</u>									
Collected: 19 Dec. 1974									
Analyzed: 13 Jan. 1975									
SS-2	1	1	10.8	1.96	35.9	655	22.8	4.56	211
	2	2	13.5 (1.2)	1.42 (.7)	43.1 (2.5)	637 (8)	21.4 (1.9)	8.51 (.5)	323 (7)
	3	3	11.5 (.2)	1.28 (.2)	35.4 (1.3)	1010 (33)	24.6 (1.4)	4.61 (.1)	250 (5)
Analyzed: 8 Apr. 1975									
SS-2	1	2	11.8 a)	1.51 a)	42.3 (1.2)	810 (36)	25.3 (1.3)	5.87 (.4)	211 (3)
	2	2	14.9 a)	1.97 a)	42.4 (2.4)	621 (20)	22.9 (.7)	9.89 (1.3)	316 (8)
	3	2	15.1 a)	1.59 a)	35.0 (.4)	759 (29)	23.6 (.6)	4.96 (.1)	245 (5)
Mean Counting Error			0.5	0.5	1.0	5.4	2.2	1.1	1.8

a) Number of runs = 1

absorption profile of Hg in air, and the others are slightly displaced ($\pm 1 \text{ cm}^{-1}$) from the absorption line. The center component is absorbed by mercury vapor, plus non-mercury decomposition products (smoke, particulates) and other thermally stable molecular species present. The slightly displaced components are absorbed only by the non-mercury background. In this system absorption due to mercury alone is measured by electronically taking the difference between the absorption of the central and displaced components. This background cancellation results in very low interferences, even in fresh tissue samples. A fully detailed description of this system is presented by Hadeishi and McLaughlin (1975).

During the present study over 5000 Hg determinations were made using this system. We have calculated the practical detection limit of Hg (limit at which results are reproducible at $\pm 5\%$) with sample weights of 5 mg, to be 0.01 ppm in lyophilized sediment and biological samples.

3. Gas-Liquid Chromatography

Samples collected during this study for chlorinated hydrocarbon determinations were analyzed at the Bodega Marine Laboratory on a Tracor MT-220 gas chromatograph equipped with a 2 meter, 4 mm I.D. glass column containing a mixture of 1.5% SP-2250/1.95% SP-2401 on 100/120 mesh Supelcon (AW-DMCS) or 3% OV-1 on 100/120 mesh Supelcopart, and 63 Ni electron capture detectors. Additionally, sidearm saponification columns were utilized to convert p,p'-DDD and p,p'-DDT, if present, to their ethylene derivatives, thus providing confirmation of their identity.

During the period of this project our laboratory participated in an intercalibration study sponsored by the U.S. Environmental Protection Agency and coordinated by the University of Washington. The intercalibration sample, divided among participating laboratories, consisted of homogenized sediment from Elliott Bay, Washington. A total of nine laboratories participated in this study. Our laboratory was designated as "A" in Table 5, reproduced from Table 2 of a report to the Environmental Protection Agency (Hom and Pavlou, 1975). The table presents the analytical results of each laboratory. The PCB concentration reported by our laboratory ($613 \pm 21 \text{ ppb}$) is equivalent to the mean value of all determinations, $609 \pm 177 \text{ ppb}$.

D. Statistical Analysis of Data

Concentrations of many metals in invertebrates may be associated with size and perhaps also with metabolic rate (Boyden, 1974). Other factors such as salinity, chemical form of the element, reproductive stage, sex, physiology, etc., could also affect uptake of trace

Table 5 Intercalibration study of chlorinated hydrocarbons in an homogenized sediment from Elliot Bay, Washington.^{a)}

Laboratory ^{b)}	Concentration in ppb ^{c)}	
	Total PCB	Total DDT
A	613 (21)	5 ^{d) e)}
B	551 (33)	18.75 ^{d)}
C	765 (46)	27.50 (4.53)
D	207 ^{d)}	NA
E	529 ^{d)}	11.6 ^{d)}
F	683 (29) 922 (53)	5.23 (1.02)
G	486 ^{d)}	64 ^{d)}
H	432 ^{d)}	68 ^{d)}
I	408 (0)	NA
Mean (all Laboratories)	609 (177)	20.32 (18)

a) From "Interlaboratory Calibration Study for Assessing Data Reliability in Chlorinated Hydrocarbon Analysis", W. Hom and S.P. Pavlou, Special Report No. 59, Feb. 1975. to the Environmental Protection Agency.

b) Laboratory designations were randomly assigned. The Bodega Bay Institute of Pollution Ecology is designated Laboratory A.

c) The error given is one standard deviation ().

d) Estimate of error not received.

e) This number did not include DDD or DDE which could not be resolved from the PCB in this sample.

elements. These factors may result in skewed, non-normal trace element distributions in populations consisting of diverse age structures and exposed to a variety of environmental conditions.

Severe time limitations and the multiple parameters that were monitored during this study made it essential to reduce the total number of samples without sacrificing accuracy of pollutant concentration estimates. To overcome the difficulties of non-normally distributed pollutant concentrations, to reduce the expected variation of sample determinations, and to limit the replicate sample number to a minimum of three, individual invertebrate and water samples were composited whenever possible.

Compositing samples is an application of the Central Limit Theorem which states that as sample size increases, the means of the samples drawn from a population of any distribution with a finite variance will approach a normal distribution (Adams, 1974). In this study 10-20 individual specimens per sample was assumed to be sufficient to assure that the distribution of the values approached normality and to reduce sample variance.

Water samples for physical - chemical characterization, and for trace element and chlorinated hydrocarbon determinations were also composited whenever possible. Owing to the hourly fluctuations of physical and chemical parameters in San Francisco Bay, water sample collections involved compositing several hourly samples into fewer, 24-hour period composites. In this manner, an estimate could be made of the average water quality and pollutant level characteristics present during the field collections. The collection techniques employed during this study did not allow compositing of surface sediment samples. Therefore, individual replicates were collected over a large area at each station to provide a best estimate of sediment pollutant levels within the two study sites.

To determine whether changes observed in pollutant concentrations during the disposal operation were different at stations within the disposal area from changes outside of the disposal area, a "group" pairing design t-test was implemented. This test is a combination of a pairing design t-test and a group comparison test. The mean concentrations observed during or just after the spoiling were subtracted from the last value obtained before disposal for each station. The differences in concentrations observed at the inner stations and the differences observed at the outer stations were subjected to the group comparison t-test. The t-test indicated whether the differences in the amount of change between inside and outside stations were significant.

Where feasible, data were subjected to analyses of variance (anova). Data from the Bodega Marine Laboratory experiments were subjected to Model I (fixed) two-way anova since the experimental conditions were determined by the investigators. Since the majority

of parameters in the field were uncontrolled, all field data were subjected to Model II (random) anova. Data for the M. edulis transplant studies were subjected to three-way anova. In this analysis, the second order interaction Error Means Square (EMS) was tested over the within subgroups EMS. The first order interaction EMS's were tested over the second order interactions EMS when significant, $p \leq 0.10$, and over significant second order interactions EMS when no affecting first order EMS's were significant. When primary interactions that affected the main effects were found, a denominator EMS for the main effects was calculated by the methods given in Simpson et al. (1960). Discussions of the general procedures of significance testing are given in Sokal and Rohlf (1969), and Snedecor (1956).

Several M. edulis transplant samples, comprising approximately 10% of the total, were lost during this study, creating special statistical problems in analyzing the data from this experiment. To minimize estimates of missing values, the data were twice subjected to three-way anova with only complete sets of data used in each analysis. One depth, one or more stations, or one collection period were excluded from each analysis. Two analyses were sufficient to include all of the data. If all the F ratios generated by these tests were used in assessing the data, the chance of rejecting a true null hypothesis (type I error) would increase to 13% from the usual 5% probability of making such an error (Woolf, 1968). Therefore, F ratios with the larger numerator degrees of freedom were chosen. When two resultant F ratios had the same numerator degrees of freedom, the F ratio with the larger denominator degrees of freedom was selected. By this method, only half of the F ratios were used in determining the significance of the data. This method was also the most sensitive for the detection of the significance of the main effects using the minimum number of estimates of missing sample values.

Assumptions for analyses of variance (anova) are that the data are normally distributed and that the variances are homogeneous. In multi-way anova the influence of one main effect on each of the other main effects is assumed to be additive. Significant interaction terms result when one or more of the main effects are not additive (i.e. the effects are not of equivalent magnitude). In these cases, conversion of the data to some equivalent scale may be necessary or interpretation of separate parts of the data may be warranted. Our data appeared to meet the above assumptions; when significant interaction terms were observed, the empirical basis of the interaction was examined carefully in the interpretation of the significance of the main effects.

E. Purge Study

As mentioned in Part II, Section B-1, those species of invertebrates which could be kept alive in the laboratory were purged for

Table 6 Purging Study. *Macoma nasuta*. Mean concentrations in parts per million dry weight of twelve elements in *Macoma nasuta* purged for zero to twelve days with stomach contents intact (A) and with stomach contents removed (B) before analysis. Mean concentrations of the elements in the stomach contents (SC) at day zero are also given. The replicate number is three with 10 individuals in each replicate. Samples were collected at Spoil Site station 2.

Days Purged:		0	3	6	12
Element	Sample				
Ag	A	7.9 (.3)	9.2 (2.3)	10.9 (3.2)	10.7 (2.8)
	B	14.2 (4.5)	9.6 (3.3)	13.6 (7.2)	13.2 (4.0)
	SC	1.9 (.6)			
As	A	13.0 (.7)	9.6 (1.6)	9.8 (1.8)	12.2 (.1)
	B	13.6 (1.3)	10.2 (1.7)	11.1 (1.2)	12.1 (.7)
	SC	10.9 (1.9)			
Cd	A	1.3 (.4)	1.9 (.5)	1.8 (.3)	2.9 (.8)
	B	1.3 (.6)	2.2 (.5)	2.0 (.3)	1.8 (.6)
	SC	2.0 (1.0)			
Cr	A	39.3 (4.2)	6.3 (.2)	4.8 (1.7)	6.4 (.3)
	B	11.7 (5.6)	6.2 (5.1)	5.5 (3.5)	4.9 (.9)
	SC	264 (7)			
Cu	A	39.3 (3.9)	27.0 (2.7)	30.8 (.9)	19.9 (1.9)
	B	50.6 (10.0)	32.1 (4.4)	30.4 (5.7)	24.5 (1.4)
	SC	52.3 (5.3)			
Fe	A	5,000 (400)	376 (33)	284 (6)	295 (13)
	B	1,480 (53)	372 (34)	301 (14)	307 (8)
	SC	33,000 (1,800)			
Hg	A	.35 (.01)	.36 (.04)	.36 (.05)	.32 (.03)
	B	.37 (.06)	.34 (.05)	.39 (.09)	.37 (.04)
	SC	.37 (.06)			
Mn	A	112 (4)	24.8 (1.4)	22.9 (1.6)	13.4 (3.4)
	B	45.8 (5.0)	28.5 (3.2)	20.0 (4.1)	18.2 (2.5)
	SC	628 (32)			
Ni	A	17.4 (2.1)	6.3 (1.4)	6.6 (1.4)	5.0 (1.4)
	B	9.5 (2.3)	6.4 (.2)	6.9 (2.6)	6.0 (1.6)
	SC	89.1 (6.5)			
Pb	A	9.1 (.8)	6.2 (.7)	7.1 (.9)	5.1 (1.9)
	B	4.5 (.9)	5.1 (1.0)	4.7 (.7)	5.0 (.9)
	SC	47.9 (5.7)			
Se	A	3.4 (.2)	3.9 (.2)	4.4 (.5)	3.8 (.3)
	B	3.9 (.3)	4.4 (.5)	3.9 (.7)	4.3 (.2)
	SC	0.8 (.2)			
Zn	A	215 (15)	225 (18)	203 (29)	216 (32)
	B	259 (67)	282 (22)	241 (35)	266 (98)
	SC	127 (8)			

Table 7 Purging study. *Macoma nasuta*. F ratios of a two-way analysis of variance, Model I, of twelve elements in *M. nasuta* (A) purged for zero to twelve days and (B) the difference between *M. nasuta* with stomach contents removed and *M. nasuta* with stomach contents intact. Degrees of freedom equal (df).

Group	(df)	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
A	3	<1	9.26***	3.92*	49.5***	24.1***	356***	<1	453***	27.1***	2.91	1.97	<1
B	1	3.45	1.2	<1	25.3***	7.07*	127***	<1	119***	4.71*	24.4***	2.42	5.84*
A x B	3	<1	<1	2.33	23.6***	1.60	130***	<1	152***	8.14*	5.41*	2.43	<1
Within subgroups 16													

* p<.05
*** p<.001

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Table 8 Benthic Invertebrates. *Pectinaria californiensis*. Purge Study. Mean concentrations in parts per million dry weight (except Fe) and one standard deviation () of twelve elements in *P. californiensis* before purging and after purging for 0, 3, 6, and 12 days. Replicate number equals three with thirty individuals per replicate. Resultant F ratios of a one-way analysis of variance between purge days are included. Degrees of freedom between groups equal three and within groups equal eight.

Days Purged	0	3	6	12	F
Element					
Ag	4.5(.7)	4.8(.6)	5.5(.3)	5.9(2.5)	.69
As	11.0(1.3)	14.1(.8)	12.4(1.9)	14.0(1.8)	2.85
Cd	3.8(1.3)	3.4(1.2)	5.3(.3)	3.9(.4)	1.40
Cr	62.5(36.8)	20.7(7.6)	10.7(8.6)	2.2(2.0)	5.81*
Cu	41.1(2.6)	36.0(.9)	40.5(3.7)	26.4(1.7)	23.36***
Fe*	.66(.2)	.19(.03)	.12(.01)	.10(.01)	18.32***
Hg	.18(.04)	.16(.01)	.22(.01)	.21(.03)	3.60
Mn	89.7(22.2)	23.3(7.6)	11.1(4.7)	6.2(1.8)	31.30***
Ni	23.3(8.3)	9.3(1.7)	7.6(2.9)	4.5(1.1)	10.14**
Pb	9.9(2.0)	6.8(1.1)	6.1(2.9)	5.8(.6)	2.96
Se	8.6(.6)	11.3(.9)	11.8(1.0)	12.6(1.4)	8.90**
Zn	117(4)	135(11)	148(.9)	151(4)	18.34***

* p<.05
** p<.01
*** p<.001

three days to remove ingested sediments to determine trace element concentrations in tissues alone. A purging study was implemented to determine the time necessary to remove the majority of ingested sediments from the alimentary systems of the clam Macoma nasuta and the polychaete Pectinaria californiensis.

Approximately 240 medium-sized (30-40 mm) M. nasuta were collected from an area adjacent to Spoil Site station 1. The clams were separated into 8 groups of 30 animals each and two groups were immediately prepared for metals analyses without purging. One of these groups was prepared with stomach contents intact ('unspitized'), while the other group was prepared for analysis after dissecting and carefully removing all sediment from stomachs and intestines ('spitized'). Two groups of clams were removed after 3, 6, and 12 days of purging. The clams were prepared in the same manner as those prepared before purging; one group with stomach contents retained and the other with stomach contents removed before analysis. Three replicate samples of 10 animals each were prepared from each of the eight groups.

The trace element concentrations in these animals are presented in Table 6. Results of a two-way analysis of variance-Model I, of these data are presented in Table 7. Neither purging or removal of stomach contents appeared to affect significantly concentrations of Ag, Hg, or Se in M. nasuta. Highly significant decreases through time occurred in As, Cr, Cu, Fe, Mn and Ni with the greatest decreases occurring in the first three days. Cadmium concentrations increased slightly but significantly over the twelve days of purging. This is contrary to findings in the Bodega Marine Laboratory uptake study in which no significant differences through time were observed even in very high ambient concentrations of Cd. Significant and highly significant differences between spitized and unspitized clams were observed in Cr, Cu, Fe, Mn, Ni, Pb and Zn indicating spitizing affected the concentrations of these metals. Significant interaction with time and spitizing in Cr, Fe, Mn, Ni and Pb indicates non-parallel rates of desorption through time between spitized and unspitized M. nasuta. In all cases desorption from unspitized clams were greater.

It was assumed that decreases in element concentrations in spitized clams indicate element loss from tissues and that differences in element concentrations between spitized and unspitized clams resulted from removal of sediments. In this investigation, the more strongly bound and stable forms of the trace elements were of primary concern. Although spitized clams contained lower concentrations of Ag, As, Cr, Cu, Fe, Mn and Ni, after three days of purging, the concentrations of these elements were very similar to element concentrations in Day 3 unspitized clams. Concentrations of Cd, Hg, Pb, Se, and Zn were also similar in both spitized and unspitized clams after three days of purging.

The greatest decreases in element concentrations occurred during the first three days of purging in spitized and unspitized clams and trace element concentrations were similar in these two groups at this point. Purging for three days was considered to be the optimal method for determining the more tightly bound metals in M. nasuta tissues.

A purging experiment was performed on M. californiensis, a species known to resist sediment purging (Nichols, 1972). Approximately 120 medium-sized (4-5 mm cephalic plate width) P. californiensis were collected from an area adjacent to Spoil Site station 1 and returned alive to the laboratory. The worms were separated into four groups of 30 animals each. One of these groups was immediately prepared for metals analysis without purging. This group served as a non-purged baseline to which the other three groups, purged for 3, 6, and 12 days could be compared. Three replicates of 10 animals each were analyzed from each group.

The results of these analyses are presented in Table 8. These data were subjected to a one-way analysis of variance, also presented in Table 8. The concentrations of the elements Ag, As, Cd, Hg and Pb were unaffected by purging whereas Se and Zn concentrations appeared to increase significantly with time.

Selenium concentrations increased approximately 30% in three days while Zn values increased by 15% during this period. These two elements may have been accumulated from the purge water or from some other source not presently known. Significant decreases with time were observed in the concentrations of Cr, Cu, Fe, Mn and Ni in P. californiensis. The greatest decrease in the concentrations of the metals Fe, Mn and Ni occurred during the first three days with smaller decreases occurring during the subsequent periods. Substantial decreases were observed in Cr and Cu concentrations in the first three days; however, the greatest decreases were observed between days 6 and 12. It is not possible to determine whether the later decreases were a result of metal desorption from tissues or the result of continued sediment purging. On the basis of the amount of fecal material found in the purging trays, at least part of these decreases are assumed to be a result of purging.

The findings of these two studies indicate that the concentrations of two elements increased slightly during purging and that some metals continued to decrease with time in P. californiensis. Three days of purging appears to be the most consistent method for estimating the concentrations of the elements in the tissues of these two species since the least amount of change was observed after this period.

F. Size Class Studies

The three species M. nasuta, P. californiensis, and M. edulis were examined for correlations between concentrations and size. For

Table 9 Size Class Study. Macoma nasuta. Mean concentrations in parts per million dry weight and one standard deviation () of twelve elements in three size classes of M. nasuta collected 22 Mar. 1975 from the Spoil Site area. The replicate number equals three with ten individuals per replicate. Resultant F ratios of a one-way analysis of variance between size classes for each element are included. Degrees of freedom between groups equal two and within groups equal 27.

Size Class (mm) a)	10-30	30-40	40-60	F
<u>Element</u>				
Ag	4.6 (.8)	9.2 (2.3)	15.7 (4.0)	12.70**
As	9.1 (1.4)	9.6 (1.6)	15.1 (1.4)	15.56**
Cd	1.5 (.6)	1.9 (.5)	2.4 (.4)	2.24
Cr	7.5 (2.4)	6.3 (.2)	6.3 (3.6)	.25
Cu	29.1 (4.3)	27.0 (2.7)	38.7 (3.8)	8.86*
Fe	897 (321)	376 (33)	494 (49)	6.28*
Hg	.24 (.01)	.36 (.04)	.47 (.07)	16.54**
Mn	38.9 (9.0)	24.8 (1.4)	25.6 (.3)	6.79*
Ni	4.4 (.9)	6.3 (1.4)	9.7 (.2)	23.29**
Pb	4.9 (1.1)	6.2 (.7)	5.2 (1.2)	1.43
Se	3.8 (.2)	3.9 (.2)	4.4 (.4)	3.78
Zn	154 (17)	225 (18)	335 (58)	18.56**

* p<.05

** p<.01

a) Size measured is the longest span across the shell.

these studies, three size classes of M. nasuta and P. californiensis were collected at the same location as was used for the purging study collections. Four size classes of M. edulis were collected at the Berkeley Pier. Each of the three size classes consisted of 60 individuals which were separated into three replicates of 20 individuals each. The animals were purged for three days and then prepared for trace element analyses.

Trace element concentrations in the three M. nasuta size classes, 10-30 mm, 30-40 mm, 38-60 mm (measurements of greatest shell lengths), are presented in Table 9. These data were subjected to a one-way analysis of variance and the results of this test are included in Table 9. Highly significant differences between the size classes were observed in the concentrations of all the elements, except Cd, Cr, Pb, and Se.

Concentrations of the elements Ag, Hg, and Zn were highest in the largest size class and lowest in the smallest size class. Arsenic, Cu and Ni concentrations were highest in the largest size class, but equivalent in the other two size classes. Concentrations of Fe and Mn

Table 10 Size Class Study. *Pectinaria californiensis*. Mean concentrations in parts per million dry weight (except Fe) and one standard deviation () of twelve elements in three size classes of *P. californiensis* collected 22 Mar. 1975 from the Spoil Site area. Replicate number equals three with 20 individuals per replicate. The resultant F ratios of a one-way analysis of variance between size classes for each element are included. Degrees of freedom between groups equal two and within groups equal six.

Size Class (mm) ^{a)}	3-4	4-5	5-8	F
Element				
Ag	3.7 (.9)	4.8 (.6)	5.6 (.8)	5.27*
As	13.2 (.7)	12.1 (.6)	8.3 (.5)	54.39***
Cd	3.6 (.9)	3.4 (.4)	3.8 (1.5)	.09
Cr	29.1 (11.9)	19.6 (3.9)	13.4 (.2)	3.56
Cu	30.7 (4.8)	37.7 (.9)	34.9 (3.9)	2.91
Fe %	0.22 (.05)	0.20 (.02)	0.14 (.02)	4.33
Hg	.15 (.01)	.18 (.00)	.20 (.01)	22.63***
Mn	27.7 (8.6)	27.7 (2.7)	19.0 (4.9)	2.14
Ni	12.7 (4.3)	8.4 (2.3)	6.5 (1.5)	3.49
Pb	6.7 (.6)	6.1 (1.7)	4.5 (1.0)	2.81
Se	11.2 (1.4)	9.4 (1.4)	6.8 (.7)	9.91**
Zn	127 (15)	132 (5)	93 (5)	15.63**

* p<.05

** p<.01

*** p<.001

a) Size measured is the cephalic plate width.

Table 11 Size Class Study. *Mytilus edulis*. Mean concentrations in parts per million dry weight and one standard deviation () of twelve elements in four size classes of *M. edulis* collected 26 Dec. 1974 from the west end of Berkeley Pier. The replicate number equals three with 10 individuals in each replicate. F ratios of a one-way analysis of variance between size classes for each element are included. Degrees of freedom between groups equal three and within groups equal eight.

Size Class (mm) ^{a)}	15-25	28-40	40-60	60-80	F
Element					
Ag	1.5 (.3)	1.2 (.4)	.7 (.6)	1.0 (.4)	1.93
As	5.2 (.4)	5.9 (1.2)	5.5 (.8)	5.8 (.2)	<1
Cd	6.0 (.3)	4.6 (.1)	5.0 (.1)	4.9 (.4)	13.70**
Cr	2.9 (1.6)	2.5 (.9)	1.7 (2.0)	1.2 (1.3)	<1
Cu	9.2 (1.2)	7.1 (.1)	6.9 (.8)	6.0 (1.7)	4.33*
Fe	115 (13)	91 (5)	94 (9)	91 (16)	3.03
Hg	.27 (.01)	.23 (.03)	.23 (.01)	.22 (.01)	9.28**
Mn	13.3 (3.5)	6.4 (1.8)	8.6 (.2)	5.9 (2.7)	5.95*
Ni	2.2 (.4)	1.6 (.3)	.6 (.2)	.5 (.2)	29.83***
Pb	4.4 (1.3)	2.9 (.7)	4.0 (1.2)	3.4 (.7)	1.17
Se	9.2 (.6)	7.3 (.7)	6.9 (1.2)	5.4 (.3)	13.33**
Zn	212 (39)	206 (58)	154 (22)	133 (34)	2.78

* p<.05

** p<.01

*** p<.001

a) Size measured is the longest span across the shell.

were greatest in the smallest size class with Mn equivalent in the two other size classes and Fe lowest in the middle size class.

The data from the P. californiensis size class study, including the results of a one-way analysis of variance, are presented in Table 10. Significant differences between size classes were observed only in the concentrations of the elements Ag, As, Hg, Se, and Zn.

The greatest concentrations of Ag and Hg were observed in the largest (5-8 mm) size classes and the lowest concentrations were found in the smallest (3-4 mm) size class. The reverse was true for Se values, while As and Zn concentrations were equivalent in the smallest and middle (4-5 mm) size classes and lowest in the largest sized animals.

Trace element concentrations in analyses of the four size classes of M. edulis collected at Berkeley Pier are presented in Table 11. These data were also subjected to one-way analysis of variance. Significant differences in the concentrations of the elements Cd, Cu, Hg, Mn, Ni, Se, and Zn were observed between two or more size classes. In each case, the highest concentrations of these elements were observed in the smallest (15-25 mm) size class. The greatest differences in trace element concentrations appeared to be between the smallest and largest size classes, with intermediate size class concentrations overlapping concentrations in these two groups.

The higher element concentrations in the smaller mussels could reflect differences in food preference, metabolic activity, water filtering rates, body surface area and/or the ability to regulate the various ions.

These studies indicate the existence of a relationship of size of these invertebrates to the concentration of a number of the elements. Therefore, the size of each of these three species must be considered when standard samples for trace element analysis are desired. In the present study, replicate samples of equally mixed sizes of M. nasuta and P. californiensis and one size class of M. edulis were analyzed so that the most reproducible estimates could be made of element concentrations within each species.

PART III. RESULTS OF FIELD STUDIES

A. Results of Trace Element Analyses of Benthic Invertebrates

Three benthic species were found in sufficient numbers at all six Spoil Site stations for monitoring of trace element concentrations during the period of study. The collection method employed during this study allowed the rapid collection of adequate numbers of the clam, Macoma nasuta (Mollusca: Pelecypoda), the worm, Pectinaria californiensis (Annelida: Polychaeta) and the sea pen, Stylatula elongata (Coelenterata: Pennatulacea). M. nasuta is both a

filter-feeder and a non-selective sediment ingester (Pohlo, 1969); P. californiensis is a selective sediment ingester (Nichols, 1974); and S. elongata is believed to be a suspension feeder.

Macoma nasuta and P. californiensis were also present in sufficient numbers at the EBMUD stations for monitoring of trace element concentrations. The amphipod Ampelisca milleri (Arthropoda: Crustacea), was found in abundance only at EBMUD stations. Other species of several phyla which were collected at these two sites were either abundant at only one or two of the sampling stations or rare. These organisms were collected for intercomparison of element concentrations at the two study sites.

1. Spoil Site

Trace element concentrations in benthic invertebrates collected at the six Spoil Site stations, through three collection periods, are presented in Tables A-1 to A-4. (Tables numbered A-1, etc. are found in Appendix A of this report.) Trace element data for the three benthic species, M. nasuta, P. californiensis and S. elongata, from the six stations and the three benthic collections were subjected to a two-way analysis of variance. The results of these tests are presented in Table 12.

No significant differences in the concentrations of the twelve elements between stations were found in M. nasuta. Significant differences were observed in the concentrations of Cu, Hg, and Ni between collections in this species. The concentrations of all three of these elements were highest during Collection II (prior to spoiling operations) and lowest during Collection I. A significant interaction was also observed for Hg concentrations indicating that the concentrations fluctuated between stations and between collection periods. Significant interaction between stations and time were also observed in the concentrations of the metals Cd, Fe, and Mn in M. nasuta (Table 12). Although there were differences in the concentrations of these metals between stations and between collections, the differences were not consistently related either to station or collection period.

Trace element concentrations were also not significantly different between stations in P. californiensis (Table 12). However, significant differences were observed in the concentrations of the elements Ag, As, Cr, Cu, Fe, Hg, and Zn between collections. Silver concentrations increased significantly in P. californiensis before spoiling operations and remained unchanged after this period (Table A-2). Concentrations of As and Fe increased significantly after Collection II to levels observed during Collection I. Only the elements Cr, Cu, Hg, and Zn showed significantly greater concentrations after spoiling operations while significant interaction between stations and

Table 12 Spoil Site Benthic Invertebrates. Results of two-way analysis of variance of the metals concentrations in *Macoma nasuta*, *Pectinaria californiensis*, and *Stylatula elongata* collected at the six Spoil Site stations and in three collections, two before and one after the spoiling operations. Degrees of freedom equal (df).

Element:	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Species												
Group (df)												
<u>Macoma</u>												
<u>nasuta</u>												
Station 5	.99	.28	.51	.52	.09	.19	1.80	.34	.86	.73	1.57	1.21
Collection 2	2.23	3.06	.75	2.18	6.89**	3.41	5.00*	2.74	9.54***	.64	1.71	.59
Sta X Coll 10	1.47	1.92	3.23**	1.46	.85	2.29*	2.43**	2.62**	1.67	.59	1.93	.70
Within subgroups 36												
<u>Pectinaria</u>												
<u>californiensis</u>												
Station 5	1.12	1.31	2.38	.19	1.16	.37	.40	.08	.11	.49	1.18	.50
Collection 2	5.05**	7.10***	1.81	3.74*	10.22**	4.97*	7.00**	2.81	3.98	.73	.50	26.74***
Sta X Coll 10	1.64	1.60	2.57**	11.22**	4.31***	4.86***	2.68**	9.22***	6.90***	8.54***	7.99***	3.28**
Within subgroups 35												
<u>Stylatula</u>												
<u>elongata</u>												
Station 5	.54	1.38	.28	.67	.68	1.09	1.01	3.53*	1.11	1.14	4.41**	.81
Collection 2	.21	5.34*	.58	2.25	1.50	20.97***	2.36	11.18**	1.36	1.13	7.83**	26.01***
Sta X Coll 10	2.68**	4.09**	2.52**	4.73**	4.06**	7.40***	.75	2.95**	1.93	2.14*	1.65	2.01
Within subgroups 35												

* p<.05

** p<.01

*** p<.001

collections was observed in the elements Cd, Fe, Mn, Ni, Pb, and Se.

Significant differences in Mn and Se concentrations between stations were observed in Stylatula elongata. Manganese and Se concentrations were significantly lower in samples collected at stations 1 and 2 during the first collection than at the other stations.

Concentrations of the elements As, Fe, Mn, Se, and Zn increased significantly through time, with the highest concentrations observed after spoiling operations (Table A-3). Significant interaction terms were revealed for the elements As, Fe, and Mn and also for Hg, Cd, Cr, Cu, and Pb (Table 12).

The trace element data for Collections II and III presented in Tables A-1 to A-3 were each subjected to a group pairing design t test to determine whether any of the differences observed in trace element concentrations were related to the spoiling operation (Tables 13-15). No significant differences were related to the disposal operation. Since no effect, or, at most, a diminished effect was expected at the Spoil Site outer stations, the disposal operation was therefore assumed not to have caused the overall increases observed.

A number of other benthic species which were either not found at all stations or not found in sufficient numbers for monitoring were also analyzed for trace element concentrations (Table A-4). The opisthobranch mollusc, Tritonia diomedea, was observed feeding on the sea pen Stylatula elongata. With the possible exception of Cd, trace element concentrations in T. diomedea did not appear to be related to element concentration in S. elongata (Table A-4). Several differences were observed in the mean trace element concentrations between bivalve species.

The species Tritonia diomedea, Glycera americana, and G. robusta, were collected at more than one station during all three collections. The data for these three species were subjected to a one-way analysis of variance and the results are presented in Table 16. The concentrations of Ni and Hg increased significantly in T. diomedea prior to Collection III and a significant, gradual increase with time was observed in the concentrations of Ag and Pb in G. americana. Glycera robusta showed significantly higher concentrations of As and Zn from Collection III and higher concentrations of Cd during Collection II.

2. EBMUD

Trace element concentrations in benthic species collected at the four EBMUD stations during the two benthic collections are presented in Tables A-5 to A-8. Only the three species M. nasuta, P. californiensis, and A. milleri were sufficiently abundant at all four stations for analysis of variance.

Table 13 Spoil Site. *Macoma nasuta*. Group Pairing Design Test. A comparison of changes in mean element concentrations observed at stations outside the spoil area before and after spoiling operations and at stations within the spoil area during this same period. $\bar{X}d_o$ = mean difference (change) at outside stations 1, 3 & 5; $\bar{X}d_i$ = mean difference at inside stations 2, 4 & 6. Degrees of freedom equal four.

Element	t	$\bar{X}d_o$	$\bar{X}d_i$	$(\bar{X}d_o - \bar{X}d_i)$
Ag	<1	-.13	-.60	.47
As	1.10	2.13	.27	1.87
Cd	<1	.00	-.57	.57
Cr	1.16	-1.10	2.33	-3.43
Cu	<1	2.23	.07	2.17
Fe	<1	39.47	-8.83	48.30
Hg	<1	.024	.008	.016
Mn	1.12	-3.37	.63	-4.00
Ni	2.19	3.03	1.40	1.63
Pb	<1	.30	-.10	.40
Se	<1	.07	-.30	.37
Zn	1.63	23.60	-19.60	43.20

No significant differences observed.

Table 14 Spoil Site. *Pectinaria californiensis* - Group Pairing Design Test.

A comparison of changes in mean element concentrations observed at stations outside the spoil area during spoiling with simultaneous changes inside the spoil area. $\bar{X}d_o$ = mean difference (change) at outside stations 1, 3, and 5; $\bar{X}d_i$ = mean difference at inside stations 2, 4, and 6. Degrees of freedom equal four.

Element	t	$\bar{X}d_o$	$\bar{X}d_i$	$(\bar{X}d_o - \bar{X}d_i)$
Ag	<1	-.20	.53	-.73
As	-1.94	-3.17	-.63	-2.5
Cd	<1	.10	.57	-.47
Cr	<1	28.10	-24.20	-3.90
Cu	<1	5.23	-3.27	-1.96
Fe	<1	-3340	-2180	-1160
Hg	<1	0.02	.04	.02
Mn	<1	-24.87	-36.77	11.90
Ni	<1	-5.90	-12.20	6.30
Pb	<1	1.53	2.07	3.60
Se	1.50	1.3	-0.50	1.80
Zn	<1	-26.63	-27.40	0.77

No significant differences observed.

Table 15 Spoil Site. Stylatula elongata. Group Pairing Design Test. A comparison of changes in the mean concentrations of each of twelve elements in Stylatula elongata at outside Spoil Site stations before and after spoiling operations and at inside Spoil Site stations during this same period. \bar{X}_d is equal to the mean difference (change) at outside stations 1, 3, and 5; \bar{X}_{d_i} is equal to the mean difference at inside stations 2, 4, and 6. The degrees of freedom equal four.

Element	t	\bar{X}_d	\bar{X}_{d_i}	$(\bar{X}_d - \bar{X}_{d_i})$
Ag	<1	-.67	-.10	-.57
As	1.06	-4.67	-.80	-3.87
Cd	<1	-.60	-.57	-0.03
Cr	<1	-1.00	-2.73	1.73
Cu	1.36	-1.00	.73	-1.70
Fe	<1	-700.9	-665.3	-35.7
Hg	2.16	-0.014	0.042	0.027
Mn	<1	-7.83	-4.40	-3.43
Ni	<1	-.70	-.83	.13
Pb	1.40	.47	1.17	-.70
Se	<1	-.37	-.30	-.07
Zn	<1	-128.7	-152.6	23.9

No significant differences observed.

Table 16 Spoil Site. Benthic Invertebrates. Results of a one-way analysis of variance of twelve elements in unequal sample sizes of Glycera americana, Glycera robusta and Tritonia diomedea collected at Spoil Site stations during three collections. Degrees of freedom equal (df).

Species	(df)	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
<u>Glycera americana</u>													
Collections	2	7.73*	1.01	1.31	1.80	2.24	.56	.02	.30	1.46	4.48*	.12	3.90
Within groups	10												
<u>Glycera robusta</u>													
Collections	2	2.38	4.40*	5.31*	2.68	2.75	.49	.32	.23	1.79	.66	.72	24.26***
Within groups	10												
<u>Tritonia diomedea</u>													
Collections	2	.49	.15	1.36	2.85	.14	.81	5.61*	.37	15.60***	.44	3.17	.76
Within groups	13												

* p<.05
 ** p<.01
 *** p<.001

Trace element concentrations in M. nasuta collected at the four EBMUD stations during the two benthic collections are presented in Table A-5. Because replicate samples of M. nasuta could not be subjected to a two-way analysis of variance with replicates (Table 17). No significant differences occurred between stations or collections in the concentrations of As, Cr, Fe, Hg, Ni, Se, or Zn. Significant differences between stations were observed in the concentrations of Ag, Cd, Cu, and Pb and between collections in Cu, Mn, and Pb. The four metals which were significantly different between stations were all found in the highest concentrations at station 3 and in the lowest concentrations at station 4. In general, the concentrations of Cu, Mn, and Pb increased at stations 1-3 and decreased at station 4. The concentrations of other elements increased or decreased randomly.

Trace element concentrations of P. californiensis collected at the four EBMUD stations are presented in Table A-6. The data were subjected to a pairing design t test for differences between Collections I and II (Table 19). Significant increases in the metals Ag, Cu, Hg, and Zn were observed between the two collection periods.

The mean concentrations and one standard deviation of the twelve elements in A. milleri collected at EBMUD stations are presented in Table A-7. The data, excluding station 4, were subjected to a two-way analysis of variance, with replication (Table 18). Significant increases in the concentrations of the elements Cr, Hg, Ni, Mn, Se, and Zn between Collections I and II and significant differences in the elements Ag, As, Cd, and Zn between stations 1, 2, and 3 were observed.

Trace element concentrations of other benthic invertebrate species collected at one or more of the EBMUD stations during the two collections are presented in Table A-8. The data for Macoma inquinata in this table and the data for M. nasuta collected during Collection II are of interest. The differences in the mean concentrations of the twelve elements observed between these two species during Collection II were subjected to a pairing design t test (Table 20). Concentrations of Ag and As were significantly higher in M. nasuta whereas concentrations of the other elements were equivalent in these two related species.

A comparison of the mean concentrations of the twelve elements in the species M. nasuta, Tapes japonica, P. californiensis, Glycera americana, and G. robusta collected at both EBMUD and Spoil Site stations is presented in Table 21. Macoma nasuta from EBMUD stations had significantly higher concentrations of the twelve elements than did Spoil Site animals, whereas there were no significant differences observed in any of the twelve elements in P. californiensis from these two areas. Concentrations between areas were similar in the polychaetes G. americana and G. robusta. The clam Tapes japonica, collected from Oakland Inner Harbor spoils material dumped at the

Table 17 EBMUD. Benthic Invertebrates. *Macoma nasuta*. Results of a two-way analysis of variance of the mean concentrations of twelve elements in *M. nasuta* collected at EBMUD stations 2, 3, and 4 during Coll. 1 (25 Nov. - 5 Dec. 1975). Degrees of freedom equal (df).

Element:	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Group (df)												
Stations (2)	14.75**	.55	6.89*	.65	12.30**	.42	1.85	3.35	.62	4.05*	.91	1.28
Collections (1)	.96	.49	4.50	2.23	18.32**	.75	.05	13.41**	.97	11.52**	1.65	1.10
Sta. X Coll. interaction	3.61	33.48**	1.68	7.06*	3.98	4.92*	4.26*	12.98**	9.49**	.72	10.66**	7.63**
within subgroups (12)												

* p<.05
** p<.01

Table 18 EBMUD. Benthic Invertebrates. Results of a two-way analysis of variance of the mean concentrations of twelve elements in *Apuliscia milleri* collected at EBMUD stations 1-3 during the two benthic collections. Degrees of freedom equal (df).

Group (df)	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Stations (2)	4.89*	6.24*	3.99*	3.79	.63	14.82	1.08	4.21	4.47	2.17	.45	20.60***
Collections (1)	3.56	1.40	.08	23.85***	1.15	11.72	22.88***	33.65*	19.65*	2.50	4.65*	26.66***
Sta. X Coll. (2)	1.07	1.60	.58	1.42	.29	9.78**	.51	11.31**	6.74*	2.24	.25	.21
(Total) (18)												

* p<.05
** p<.01
*** p<.001

Table 19 EBMUD. Benthic Invertebrates. Mean concentrations in parts per million dry weight of twelve elements in *Pectinaria californiensis* collected at four EBMUD stations during Collection I (5 Dec. 1974) and Collection II (13 Feb. 1975). Resultant t ratios of a pairing design test for each element between collections are included. Degrees of freedom equal seven.

Mean Concentrations:	\bar{x}_1	\bar{x}_2	t ratio
Element			
Ag	2.51	4.24	-2.43*
As	10.03	12.12	-1.42
Cd	2.31	3.74	-1.98
Cr	18.99	24.30	-0.71
Cu	19.43	33.17	-4.64**
Fe	2,787	2,556	0.36
Hg	.14	.23	-3.44*
Mn	17.43	20.24	-0.77
Ni	5.20	9.79	-3.97**
Pb	5.50	6.45	-0.79
Se	11.42	13.49	-1.30
Zn	87.47	170.01	-4.85**

* p<.05

** p<.01

Table 20 EBMUD. Benthic Invertebrates. Results of a pairing design t test of the differences in the mean concentrations in parts per million dry weight of twelve elements in *Macoma nasuta* and *M. inquinata* collected at the EBMUD stations during Collection II. Degrees of freedom equal 10.

Element	<i>M. inquinata</i>	<i>M. nasuta</i>	t
Ag	7.2	19.9	3.73**
As	11.6	21.5	3.08*
Cd	1.8	1.5	.76
Cr	19.4	14.8	1.72
Cu	69.9	96.7	1.05
Fe	1373	942	1.51
Hg	.68	.78	1.90
Mn	84.2	67.9	.98
Ni	8.9	9.4	.45
Pb	12.9	13.4	.32
Se	6.2	5.4	.69
Zn	404	417	.18

* p<.05

** p<.01

Table 21 Spoil Site - ERMUD. Comparison of the mean concentrations and one standard deviation () of twelve elements in five benthic invertebrates collected at ERMUD and Spoil Site stations during the period of study. Values are expressed as ppm dry weight. Replicate number of samples (n) is given. Each replicate consisted of 5 to 20 individuals.

Species	Stations: n	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
<i>Macoma nasuta</i>	S.S.: 18	12.9(2.3)	16.3(1.5)	1.4(.4)	6.1(2.4)	38.6(4.1)	688(135)	.39(.05)	23.6(3.5)	8.0(1.4)	5.3(.7)	4.2(.5)	273(29)
	ERMUD: 8	19.4(4.8)	21.0(3.7)	1.9(.6)	11.9(5.7)	83.3(29)	884(279)	.76(.11)	51.3(28)	8.9(1.8)	11.9(2.3)	4.8(1.4)	377(96)
<i>Pectinaria californiensis</i>	S.S.: 18	3.7(.7)	11.4(1.4)	3.2(.7)	29.8(21)	30.3(5.7)	3800(2100)	.16(.02)	46.3(29)	11.6(6.3)	7.6(2.4)	9.6(1.6)	104(31)
	ERMUD: 7	3.3(1.4)	17.9(12)	3.0(1.2)	22.0(12)	21.0(11)	2800(800)	.18(.05)	17.7(6.5)	7.1(2.9)	5.9(1.8)	12.4(2.5)	127(54)
<i>Tapes japonica a)</i>	S.S.: 2	9.6(.8)	24.1(1.1)	6.4(1.6)	10.2(5.9)	13.0(.5)	1395(932)	.90(.18)	20.2(7.2)	11.2(3.3)	16.9(6.2)	9.6(.1)	172(76)
	ERMUD: 2	4.4(1.5)	6.1(1.1)	4.1(.5)	9.3(9.4)	22.0(.1)	1087(913)	.26(.02)	17.3(10)	7.1(1.8)	3.9(1.5)	6.9(.9)	125(21)
<i>Glycyca americana</i>	S.S.: 11	3.0(1.0)	16.3(1.5)	5.7(1.5)	5.7(1.5)	24.0(8.2)	772(182)	.10(.00)	7.7(.6)	2.1(.4)	4.0(1.0)	5.2(.1)	200(25)
	ERMUD: 2	2.6(.3)	21.6(8.9)	3.2(.6)	5.1(1.6)	24.6(4.2)	880(277)	.12(.03)	9.5(.6)	2.4(1.1)	4.0(.6)	5.7(1.1)	217(47)
<i>Glycyca robusta</i>	S.S.: 12	3.2(.9)	30.7(17)	3.3(1.5)	4.3(2.5)	51.3(20)	738(101)	.10(.01)	7.2(.4)	2.4(1.0)	4.8(.6)	5.4(.2)	192(56)
	ERMUD: 1	3.6	12.7	1.9	.7	13.9	831	.08	9.3	1.9	6.8	4.8	222

a) *T. japonica* did not occur at the spoil site. The samples collected there were carried in with the dredge spoil material.

Spoil Site, had higher concentrations of the elements Ag, As, Hg, and Pb and a lower concentration of Cu than did T. japonica from EBMUD stations.

B. Mytilus edulis Transplant Study

1. Description of Transplant Experiments

The mussels, M. edulis, were transplanted to three depths at the six Spoil Site stations to determine whether disposal of dredged material containing elevated levels of trace elements and chlorinated hydrocarbons would result in elevated concentrations of these compounds in a filter-feeding invertebrate. Mytilus edulis has been used in other studies to monitor pollutant levels (Schulz-Baldes, 1973). In the present study the relationships of pollutant concentrations in this species with pollutant concentrations in water and suspended particulates were determined.

Mussels were also transplanted to three depths at the four EBMUD stations to compare possible trace element and chlorinated hydrocarbon accumulation from treated sewage effluent with possible accumulation of these compounds from dredge spoil disposal operations at the Spoil Site.

The mussels used in these transplant experiments were collected from the west end of Berkeley Pier. Berkeley Pier mussels were monitored during the course of this study to determine fluctuations of pollutant levels in a reference population.

2. Collection and Preparation of M. edulis Transplants

Results given by Anderlini et al. (1975) indicated that M. edulis from outside of San Francisco Bay responded differently to environmental fluctuations in trace element concentrations than did M. edulis obtained locally. Mussels collected from central San Francisco Bay, in proximity to both the EBMUD and the experimental Spoil Site, were therefore assessed to be suitable for these transplant experiments.

Approximately 6,800 M. edulis were collected from the west end of Berkeley Pier for use in subsequent experiments. Mussels were washed clean of adhering sediment and epifauna in the field and returned alive to the laboratory.

In the laboratory, sixty individuals were selected for baseline value determinations, purged, and prepared for analyses as described in Part II, Section B. Another 240 individuals were grouped by size into four distinct size classes (15-25, 28-40, 40-60 mm) to determine the relationship of size (shell length) and metal concentration (see Boyden, 1974).

Approximately 6,500 mussels between 20-40 mm in length were selected for the transplant study. These mussels were divided into 108 groups of 60 mussels each and each group was placed in a 50 cm x 15 cm, 1.0 mm mesh Nytex bag. Plywood frames were constructed to hold these Nytex bags and for vertical "attachment" to the subsurface buoys at each of three depths (Figure 2).

One frame with four bags per frame was placed at each of the three depths at each of the subsurface stations. Transplant bags from each station and depth were collected according to the schedule listed in Table 1. Transplanted mussels collected during this study were returned to the laboratory and prepared for chlorinated hydrocarbon and trace element analyses as described in Part II, Section B.

3. Results of Trace Element Analyses of M. edulis Transplants

a. Spoil Site

Four transplant collections were made at each of the six Spoil Site stations; two before and two after spoiling operations. The second pre-spoiling collection was conducted one week before spoiling operations commenced and the first post-spoiling collection was conducted two days after spoiling ceased. The results of the trace element analyses of the four Spoil Site M. edulis transplant collections are presented in Table A-9.

Since some of the transplant bags were lost during the period of study, the data were subjected twice to a three-way analysis of variance (Model II) to avoid estimating missing samples. The results of these three-way analyses of variance are presented in Tables 22 and 23. A discussion of this procedure is presented in Part II, Section D.

No significant differences were observed between stations in any of the elements. Lead concentrations were not significantly different between collections or depths. Significant differences between depths were observed in the concentrations of Cd, Cu, Fe, Hg, Mn and Ni. In general, the concentrations of these elements were significantly lower at the middle depth.

The concentrations of all the elements except Ag and Pb were significantly different between collections. The concentrations of As, Ni, and Zn were greatest during Collection IV and lowest during Collection II. Cadmium, Fe, and Mn concentrations were also greatest during Collection IV, but were lowest during Collection III.

No significant interactions were observed between depths and collection periods or between stations and depths in any element concentrations except Cr. In the case of Cr values, significant interaction between depth and collection period resulted from apparently random fluctuating mean concentrations at the three depths during the four collections (Table A-9).

Table 22 Spoil Site, Transplant Study. F ratios of three-way analysis of variance Model II. Comparison of metals concentrations in M. edulis between stations 1, 2, 4 and 6, at three depths through four collections. Denominator degrees of freedom (ddf) for the main effects are included.

	df	Ag	As	Cd	Cr	Cu	Fe
		ddf	ddf	ddf	ddf	ddf	ddf
Stations	3	<1	<1	2.79*	<1	3.34*	<1
Depths	2	<1	2.97	5.40**	<1	4.57*	5.38*
Collections	3	<1	48.44***	18.73***	2.50	11.75**	11.91***
Sta. X Depths	6	<1	1.97	<1	<1	<1	<1
Sta. X Coll.	9	<1	2.84	1.84	<1	1.54	<1
Depth X Coll.	6	<1	1.02	1.83	2.95*	1.16	<1
Sta. X Depth X Coll.	18	2.16	1.36	1.24	<1	1.64	5.55**
Within subgroups	96						

	Hg	Mn	Ni	Pb	Se	Zn
Stations	<1	2.20	1.43	<1	<1	5.60**
Depths	4.5*	4.64*	3.42	<1	1.91	<1
Collections	10.0**	5.98*	14.26***	<1	4.80*	11.71**
Sta. X Depths	1.5	<1	<1	<1	2.28	1.65
Sta. X Coll.	2.0	2.33	1.33	<1	1.82	1.82
Depth X Coll.	2.0	<1	1.40	<1	2.72*	2.36
Sta. X Depth X Coll.	3.00**	1.10	2.31**	1.60	<1	<1
Within subgroups	96					

* p<.05
 ** p<.01
 *** p<.001

Table 23 Spoil Site. Transplant Study. F ratios of three-way analysis of variance, Model II. Comparison of metals concentrations in M. edulis at all stations and middle and bottom depths through four collections. Denominator degrees of freedom (ddf) for the main effects are included.

	df	Ag	ddf	As	ddf	Cd	ddf	Cr	ddf	Cu	ddf	Fe	ddf
Stations	5	1.33	15	1.41	15	<1	-	<1	96	1.34	.5	1.07	6.0
Depths	1	<1	15	<1	15	<1	-	2.19	96	2.65	2.5	1.29	4.1
Collections	3	3.00	15	20.11**	15	3.45	3.2	6.48**	96	3.96	2.7	10.03**	15
Sta. X Depth	5	1.33		<1		2.58		<1		<1		2.57	
Sta. X Coll.	15	1.67		<1		1.22		1.40		<1		1.47	
Depth X Coll.	3	<1		<1		4.87		<1		3.84*		1.19	
Sta. X Depth X Coll.	15	1.65		2.67**		1.63		1.27		2.10*		3.57**	
Within subgroups	96												

	Hg	Mn	Ni	Pb	Se	Zn							
Stations	5	1.0	-	4.53**	96	1.4	15	<1	-	1.89	15	<1	-
Depths	1	<1	-	<1	3.3	<1	2.8	<1	-	<1	15	<1	-
Collections	3	2.8	4.24	2.48	3.3	5.2	2.8	<1	-	9.56***	15	10.96*	3.4
Sta. X Depth	5	2.00		1.03		<1		1.05		1.11		3.18*	
Sta. X Coll.	15	1.66		1.03		<1		<1		1.0		1.32	
Depth X Coll.	3	2.66		4.90**		4.0*		<1		1.9		3.40*	
Sta. X Depth X Coll.	15	3.32**		<1		2.51**		1.63		2.66**		1.53	
Within subgroups	96												

* p<.05
 ** p<.01
 *** p<.001

Significant second order interactions between stations, depths, and collections were observed for the elements Fe, Hg, and Ni. Concentrations of the majority of the trace elements increased during the month after spoiling. These data were subjected to a group pairing design t test. The changes in trace element concentrations which occurred during spoiling in transplants at the inner stations were compared with changes which occurred in mussels from the outer stations (Table 24). Only the concentration changes of Se and Zn were significant. Concentrations of Zn decreased at the inner stations, while concentrations at the outer stations increased. Selenium concentrations increased significantly less at inner stations than at outer stations. Therefore, spoiling operations apparently decreased the availability of Se and Zn to inner station transplants.

Table 24 Spoil Site Transplant Study. *Mytilus edulis*. Group Pairing Design Test. A comparison of changes in mean trace element concentrations observed at stations outside of the spoil area before and after spoiling and at stations within the spoil area during this same period. \bar{X}_d_o = mean difference (change) observed at outside stations 1, 3 and 5; \bar{X}_d_i = mean difference observed at inside stations 2, 4 and 6. Degrees of freedom equal fourteen.

Element	t	\bar{X}_d_o	\bar{X}_d_i	($\bar{X}_d_o - \bar{X}_d_i$)
Ag	1.29	.40	.10	.30
As	1.07	-1.90	-1.41	-.49
Cd	<1	.44	.12	.32
Cr	1.21	.00	.75	-.75
Cu	1.65	-1.76	-.58	-1.18
Fe	1.28	34.80	49.62	-14.82
Hg	<1	-.01	.01	-.02
Mn	<1	.73	.67	.06
Ni	1.35	-.53	.02	-.55
Pb	<1	-.37	.09	-.46
Se	2.32*	-.99	-.02	-.96
Zn	2.43*	-19.57	6.56	-26.13

* $p < .05$

b. EBMUD

Mussels, transplanted to three depths at the four EBMUD stations (Figure 1), were removed during three collections (Table 1). Trace element concentrations of these mussels are presented in Table A-10.

The data were subjected to a three-way analysis of variance (Model II).

Since transplant bags were also lost at this site, the data were subjected to two separate three-way analyses of variance. The first group of data included transplants collected at all stations and depths during the first two collections only (Table 25). The second three-way analysis of variance of transplant concentrations between stations 1, 2, and 4 at all depths and during all three collections are presented in Table 26.

No significant differences were observed in the concentration of any element between stations or between depths. Significant continuous increases with time were observed in the elements As, Cu, Hg, Mn, Ni, Pb, Se, and Zn. The highest concentrations of these elements occurred during Collection III and the lowest concentrations were observed in Collection I. A continuous increase in trace element concentrations occurred through time at this site. The increases may be related to an increased influx of fresh water during this period or to a number of other chemical or physical influences. Significant first order interactions between stations and collections and between depths and collections were observed in Ag, Cd, and Hg concentrations. A significant interaction between stations and depths was observed in Ag. Significant second order interactions were observed in the concentrations of As and Zn.

To determine to what extent natural fluctuations of the elements were in correlation with concentrations observed at Spoil Site and EBMUD stations, mussels were collected at Berkeley Pier during four periods (Table 1). The results of trace element analyses of these "control" mussels are presented in Table A-11. A gradual increase was observed in the concentrations of all the elements except Ag and Cr. Berkeley Pier mussel trace element concentrations were compared with the grand means of trace element concentrations in Spoil Site and EBMUD transplants collected from the shallowest depth at all stations during each collection (Table 27). Although the collection periods varied between sites, gradual increases were also observed in the concentrations of all elements in Spoil Site and EBMUD transplants.

The magnitude of increases in element concentrations observed at all three sites appeared to be equivalent and probably represented a general increase in trace element concentrations in Central Bay waters during this period. Increases in the element concentrations in Spoil Site mussels were no greater than increases in Berkeley Pier or EBMUD mussels and, therefore, are not considered to be related to spoiling operations.

4. Results of Chlorinated Hydrocarbon Analyses

Chlorinated hydrocarbon concentrations in M. edulis transplants

Table 25 Enslin, Transplant Study. F ratios of three-way analysis of variance Model II. Comparison of metals concentrations in transplanted *M. edulis* between stations 1-4, three depths and collections 1 and II. Denominator degrees of freedom (ddf) for the main effects are included.

	df	Ag	As	Cd	Cr	Cu	Fe	
		ddf	ddf	ddf	ddf	ddf	ddf	ddf
Stations	3	<1	<1	<1	2.63	<1	1.66	48
Depths	2	<1	4.02*	3.29	<1	2.88	1.13	48
Collections	1	2.87	86.30***	1.23	<1	8.06**	<1	48
Sta. X Depth	6	2.86*	<1	1.19	<1	<1	1.65	
Sta. X Coll.	3	<1	<1	<1	<1	<1	1.11	
Depth X Coll.	2	3.21	1.55	2.04	1.69	<1	1.66	
Sta. X Depth X Coll.	6	1.37	<1	2.73*	<1	<1	<1	
Within subgroups	48							

	Hg	Mn	Ni	Pb	Se	Zn
	ddf	ddf	ddf	ddf	ddf	ddf
Stations	<1	1.00	1.89	<1	1.24	1.89
Depths	<1	2.01	<1	<1	<1	3.41
Collections	6.04	1.79	21.68***	2.15	9.17	67.92**
Sta. X Depth	2.18	1.22	1.86	<1	2.53	1.56
Sta. X Coll.	4.15*	<1	<1	<1	<1	1.03
Depth X Coll.	6.99**	<1	1.03	2.15	5.15*	<1
Sta. X Depth X Coll.	1.23	1.02	1.21	1.27	<1	<1
Within subgroups	48					

* p<.05
 ** p<.01
 *** p<.001

Table 26 EBBUD. Transplant Study. F ratios of three-way analysis of variance, model II. Comparison of metals concentrations in *M. edulis* between stations 1, 2 and 4 at three depths through three collection periods. Denominator degrees of freedom (ddf) for the main effects are included.

	Au	As	Cd	Cr	Cu	Fe
	df	ddf	ddf	ddf	ddf	ddf
Stations	2	7.50**	59	<1	-	<1
Depths	2	<1	-	<1	<1	<1
Collections	2	<1	-	1.83	59	<1
Sta. x Depth	4	<1	-	18.07**	54	2.42
Sta. x Coll.	4	1.57	<1	2.82	54	2.42
Depth x Coll.	4	3.38*	<1	2.55	<1	2.78*
Sta. x Depth x Coll.	4	2.71*	<1	<1	<1	1.66
Within subgroups	54	1.26	2.18*	3.84**	1.18	1.84
				1.50	1.40	<1
	Hg	Mn	Ni	Pb	Se	Zn
	df	ddf	ddf	ddf	ddf	ddf
Stations	2	<1	-	<1	-	<1
Depths	2	<1	-	<1	<1	<1
Collections	2	20.00**	5.5	7.43**	54	12.61**
Sta. x Depth	4	1.31	2.54	2.19	<1	9.48**
Sta. x Coll.	4	4.20**	<1	<1	<1	<1
Depth x Coll.	4	6.03**	<1	1.97	<1	1.04
Sta. x Depth x Coll.	4	1.21	<1	1.46	1.93	1.50
Within subgroups	54				3.58	2.51*

* p<.05
 ** p<.01
 *** p<.001

Table 27 Transplant Study. Mean and grand mean concentrations in parts per million dry weight and one standard deviation () of ten elements in *Mytilus edulis* from Berkeley Pier and in the top transplants at the Spoil Site and EBMUD stations. Berkeley Pier values are means and Spoil Site and EBMUD values are grand means. Sample size (n) is included.

Collections:			I	II	III	IV
Element	Location	(n)				
As	BP	3	3.6(1.0)	5.9(1.2)	7.6(.6)	6.1(1.1)
	SS	4	7.6(.4)	7.5(.3)	8.6(.2)	10.1(.5)
	EBMUD	4	5.5(.1)	7.9(.4)	8.2(.1)	a)
Cd	BP	3	4.2(.0)	4.6(.1)	6.8(1.0)	7.2(.5)
	SS	4	5.2(.2)	5.3(.2)	5.2(.1)	5.9(.1)
	EBMUD	4	5.4(.2)	5.5(.2)	6.3(.1)	—
Cu	BP	3	5.6(1.2)	7.1(.1)	8.8(.4)	10.6(.5)
	SS	4	9.5(.4)	8.2(.7)	9.5(.2)	11.4(1.1)
	EBMUD	4	8.8(1.1)	10.4(.4)	13.0(1.8)	—
Fe	BP	3	83(9)	91(5)	133(25)	134(6)
	SS	4	141(6)	141(5)	98(7)	196(13)
	EBMUD	4	117(11)	104(5)	144(4)	—
Hg	BP	3	.25(.01)	.23(.03)	.49(.04)	.52(.01)
	SS	4	.33(.01)	.31(.01)	.31(.01)	.32(.02)
	EBMUD	4	.35(.02)	.43(.02)	.50(.01)	—
Mn	BP	3	4.3(2.0)	6.4(1.8)	10.0(2.4)	9.5(1.9)
	SS	4	6.7(.8)	7.7(.5)	7.1(.6)	11.3(1.8)
	EBMUD	4	6.9(.7)	8.1(1.1)	9.9(1.2)	—
Ni	BP	3	.3(.5)	1.6(.3)	3.0(.7)	3.6(.4)
	SS	4	2.1(.2)	1.9(.2)	2.0(.1)	3.2(.5)
	EBMUD	4	1.7(.2)	2.6(.2)	3.2(.3)	—
Pb	BP	3	3.2(1.7)	2.9(.7)	4.8(.9)	5.6(.4)
	SS	4	3.7(.1)	3.5(.5)	4.2(.4)	4.3(.3)
	EBMUD	4	3.9(.2)	4.3(.3)	6.6(.6)	—
Se	BP	3	5.9(.2)	7.3(.7)	9.9(.4)	9.4(.9)
	SS	4	9.1(.2)	9.2(.2)	9.6(.4)	11.0(.2)
	EBMUD	4	6.4(.1)	7.5(.1)	8.5(.4)	—
Zn	BP	3	127(6)	206(58)	245(88)	275(21)
	SS	4	219(6)	212(6)	214(13)	283(13)
	EBMUD	4	224(7)	285(5)	386(35)	—

a) Only three transplant collections were made at EBMUD stations.

collected at the six Spoil Site and four EBMUD stations during all collections are presented in Tables A-12 and A-13 respectively. The mean concentrations and one standard deviation of the chlorinated hydrocarbons observed in M. edulis from all Spoil Site stations during the four collections are presented in Table 28.

Concentrations of p,p'-DDE decreased through time and differed between stations. The highest concentrations of this compound were observed at station 6 and the lowest at station 5. No differences were observed in p,p'-DDD or PCB levels through time or between stations during the period of study.

Results of a group pairing t test, presented in Table 29, reveal a highly significant difference between changes in p,p'-DDE concentrations at inner and outer stations. While there was a gradual decrease in the concentration of this compound through time at all stations, concentrations in M. edulis from outside stations decreased significantly more than in mussels from inner stations after the spoiling operation. Concentrations of p,p'-DDE were equivalent at inner and outer stations during pre-spoiling collection and again during collection IV, one month after the spoiling operation (Table 28). The greater decrease observed at outer stations immediately after the disposal period suggests that inner station mussels were exposed to higher concentrations of this compound during spoiling operations and that the desorption of p,p'-DDE observed in outer station mussels was prevented in inner station animals. Chlorinated hydrocarbon concentrations in EBMUD and Berkeley Pier mussels are presented in Tables 30 and 31. The decreasing concentrations of p,p'-DDE observed in Spoil Site outer stations, Berkeley Pier, and EBMUD during this period and the lack of a similar decrease in p,p'-DDE concentration in inner Spoil Site mussels immediately after the disposal period further indicate that the spoiling operation increased p,p'-DDE availability.

The PCB profiles of mussels analyzed in this study most closely resembled Aroclor 1254. Chromatographic profiles of the Aroclor 1254 standard and extracts of mussel tissues were similar. DDT compounds generally contributed less than 15% of the total chlorinated hydrocarbon residues.

Polychlorinated biphenyl and DDT concentrations in M. edulis from San Francisco Bay are compared in Table 32 with PCB and DDT concentrations in Mytilus spp. from other localities. These values indicate the relative concentrations of these pollutants in San Francisco Bay. Chlorinated hydrocarbon concentrations in mussels from the EBMUD were comparable to those in other estuarine and marine regions of the world subject to industrial wastes and effluents.

Table 28 Spoil Site. Transplant Study. Mean concentrations in parts per billion wet weight of chlorinated hydrocarbons observed in transplanted *Mytilus edulis* collected at three depths at the six Spoil Site stations during four collections: I (16 Jan. 1975), II (26 Jan. 1975), III (6 Feb. 1975), and IV (17 Mar. 1975). The replicate sample size, n, is given for each depth and collection. Each replicate consisted of ten individuals.

Compound:			p,p'-DDE	p,p'-DDD	PCB Aroclor 1254
Collection	Level	n			
I	Top	4	6.2 (1.5)	3.4 (.8)	60 (15)
	Middle	5	5.1 (.8)	5.1 (.8)	108 (16)
	Bottom	6	5.4 (.4)	2.8 (.3)	50 (5)
II	Top	4	7.0 (.8)	3.0 (.2)	51 (10)
	Middle	6	5.3 (1.0)	4.5 (.9)	81 (14)
	Bottom	5	5.6 (2.1)	2.9 (1.1)	46 (20)
III	Top	4	5.8 (1.2)	5.4 (1.2)	81 (18)
	Middle	6	3.8 (1.2)	3.5 (.9)	68 (11)
	Bottom	6	3.2 (1.2)	4.0 (1.5)	67 (16)
IV	Top	4	3.4 (1.0)	2.9 (.8)	82 (21)
	Middle	5	2.2 (.7)	2.7 (1.0)	70 (25)
	Bottom	5	2.0 (1.5)	2.2 (1.2)	60 (20)

Table 29 Spoil Site. Transplant Study. Resultant t values of a group pairing design test of the changes in the mean chlorinated hydrocarbon concentrations in *Mytilus edulis* at stations outside the spoil area before and after spoiling operations and at stations within the spoil area during this same period. \bar{X}_d = mean difference in ppb wet weight at outside stations 1, 3, and 5; \bar{X}_{d_i} = mean difference at inside stations 2, 4, and 6. Degrees of freedom equal 13.

Compound	t	\bar{X}_{d_i}	\bar{X}_{d_o}	$(\bar{X}_{d_i} - \bar{X}_{d_o})$
p,p'-DDE	-4.91***	.57	3.35	-2.78
p,p'-DDD	-1.92	-1.37	.26	-1.62
PCB (Aroclor 1254)	-1.70	-19.56	1.07	-20.62

*** p<.001

Table 30 ERMUD. Transplant Study. Mean concentrations and one standard deviation () of chlorinated hydrocarbons in transplanted *Mytilus edulis* collected at three depths at the four ERMUD stations during three collections: I (10 Jan. 1975), II (11 Feb. 1975) and III (18 Mar. 1975). Values are expressed as PPB wet weight. The number of stations sampled is equal to the number of replicate samples in each collection at each depth.

Collection	Level	Compound			PCB Aroclor-1254
		P,p'-DDE	P,p'-DDD		
I	Top	5.8 (1.1)	3.6 (1.0)		79 (20)
	Middle	6.8 (1.2)	3.8 (.5)		78 (4)
	Bottom	6.1 (1.2)	3.4 (.3)		74 (8)
II	Top	4.4 (.6)	8.6 (.2)		102 (17)
	Middle	2.9 (2.9)	5.1 (.5)		79 (8)
	Bottom	2.2 (1.1)	4.4 (1.2)		69 (15)
III a)	Top	1.1 (.5)	8.2 (3.8)		138 (17)
	Middle	1.6 (.9)	8.0 (2.3)		136 (12)
	Bottom	2.1 (1.2)	8.7 (1.3)		167 (45)

a) three stations in collection III.

Table 11 Transplant Study, Berkeley Pier. Concentrations of chlorinated hydrocarbons in *Mytilus* adults collected at the Berkeley Pier control station at three collection times. Values are expressed as ppb wet weight.

Date	p,p'-DDE	p,p'-DDD	PCB Aroclor 1254	Shell length range (mm)
21 Dec. 1974	7.3	3.6	64	30.7-39.9
2 Feb. 1975	4.5	4.5	70	33.0-42.8
8 Feb. 1975	2.4	4.7	67	29.7-40.4
Mean:	4.7 (2.5)	4.3 (0.6)	67 (3)	

Table 12 Transplant study. Concentrations of chlorinated hydrocarbons in *Mytilus* spp. from various global areas. Values are expressed as ppb wet weight.

Reference	Year sampled	Area sampled	Total ¹⁾	
			DDE	PCB
Jensen et al., 1969	1966	W. Coast of Sweden		11-130
	1968	Baltic Sea		8-57
		Archipelago of Stockholm		
Koeman et al., 1969	1968	Rhine R.		1200
		Mouth of Rhine R.		200
de Lappe et al., 1972	1972	Medit. coast of France		110-1920
Baluja et al., 1973	1972	Medit. coast of Spain	25-180	58-1360
		Atlantic coast of Spain	12-17	100-670
de Lappe et al.,	1971	S. Calif. Bight and Bodega Bay, Calif.	19-4200	14-520
	1974	S. Calif. Bight	6.7-1270	5.8-370
This study	1975	San Francisco Bay	3-15	61-147

1) Total DDT is the sum of the DDE, DDD and DDT compounds observed.

C. Results of Analyses of Benthic Sediments and Settled
Particulates

1. Particle Size Analyses

Particle size analyses of benthic sediments and settled particulates collected at Spoil Site and EBMUD stations during this study are presented in Tables A-14 to A-21. The mean percent of particle sizes in sediments collected from the hoppers of the dredge ship CHESTER HARDING before each of the seven experimental disposals at the Spoil Site are included in Table A-15. Means percents of particle sizes of sediments collected prior to dredging in Oakland Inner Harbor and at the impact center of the experimental Spoil Site after spoiling are also included in Table A-15.

a. Spoil Site

Three benthic sediment collections were made at the Spoil Site stations; two prior to and one following the spoiling operation. Particle size distributions in the four sediment core layers collected at three of the Spoil Site stations during the first collection are presented in Table A-14. No differences were observed in the mean percentage of sand, silt and clays among the four depths. Sand was the most abundant component of these sediments, coarse clays the least.

Particle size distributions in surface (0-2 cm) sediments collected before and after the spoiling operation are presented in Table A-15. There were no major differences observed in the particle size composition of the sediments before or after the spoiling operation. The mean percentages of sand, silt, and clays in sediments collected at the Oakland Inner Harbor dredge zone were similar to sediments disposed of at the Spoil Site. After the disposal, the percentage of silts and clays in the spoiled material at the center of the Spoil Site decreased by approximately 50%. This may have been a result of removal and redistribution of the finer particles by strong bottom currents observed in this area. This possible redistribution of spoiled sediments did not appear to affect the particle size distribution of surface sediments at any of the surrounding Spoil Site stations and may have occurred only on the surface of the spoiled sediments.

Particle size distributions in settled particulates collected at Spoil Site stations before and after the experimental spoiling operation are presented in Table A-16. No significant differences were observed in particle size distribution immediately after spoiling. However, one month later consistent differences were observed in the percent of sand and fine clays in settled particulates. The mean percentage of sand increased by approximately 50% while fine clays

decreased by about 30%. The changes in percent sand and fine clays may be the result of natural inputs of river-borne sediments or the result of the stirring action of gale force winds which occurred between the last two collections.

The residence time of collecting tubes and the particulate deposition rate in $\text{mg}/\text{cm}^2/\text{day}$ are presented in Table A-17. Estimated sedimentation rates were slightly higher at stations 1, 3, 4, and 5 after the disposal than before and very much greater at station 2 after this operation. This large increase at station 2 was assumed to be the result of observed spoiling almost directly over this station.

The grand mean percentages of sand, silt, and clays in surface sediments collected during the two pre-spoiling and the one post-spoiling collections were compared with the mean percentages of particle fractions of settled particulates also collected before and after the spoiling operation (Table 33). Particle size distributions of surface sediments and settled particulates were significantly different. The greatest differences appeared to be in the percentage of sand and clays. The percent sand in surface sediments was 5 times greater than in settled particulates before spoiling and decreased to 2 times higher after the experimental disposals. This increased percentage of sand in settled particulates after spoiling may reflect the high sand content of the dredge spoil material.

b. EBMUD

A comparison of the mean percentages of sand, silt, and clays with depth is presented in Table A-18. The percentages of silts and coarse and fine clays were not greatly different with depth or between stations. However, the percentage of sand in the 30-60 cm layers was greater than the percent sand in the 0-2 cm layer at all four stations. Sand content of sediments from the 10-30 cm layers at stations 1, 2 and 4 were also greater than in the 0-2, and 2-10 cm layers at these stations. The sand content of sediments collected from 0-2, 2-10, and 10-30 cm depths at EBMUD station 4 was also greater than at other stations at these depths (Table A-18). Silts and clays apparently predominated in the 0-2 and 2-10 cm layers of sediments at stations near the EBMUD outfall and decrease in percentage away from the outfall. This trend of increasing sand content and decreasing silts and clays is suggested by the data for the four stations at 0-2, 2-10, and 10-30 cm depths. This phenomena was probably due to the proximity of the very large EBMUD outfall in this area and the proximity of station 4 to the dredged, channel entrance to Oakland Outer Harbor (Figure 1).

With the exception of a slight decrease in the percentage of sand in sediments from stations 1, 2, and 4 during the second collection,

C. Results of Analyses of Benthic Sediments and Settled
Particulates

1. Particle Size Analyses

Particle size analyses of benthic sediments and settled particulates collected at Spoil Site and EBMUD stations during this study are presented in Tables A-14 to A-21. The mean percent of particle sizes in sediments collected from the hoppers of the dredge ship CHESTER HARDING before each of the seven experimental disposals at the Spoil Site are included in Table A-15. Means percents of particle sizes of sediments collected prior to dredging in Oakland Inner Harbor and at the impact center of the experimental Spoil Site after spoiling are also included in Table A-15.

a. Spoil Site

Three benthic sediment collections were made at the Spoil Site stations; two prior to and one following the spoiling operation. Particle size distributions in the four sediment core layers collected at three of the Spoil Site stations during the first collection are presented in Table A-14. No differences were observed in the mean percentage of sand, silt and clays among the four depths. Sand was the most abundant component of these sediments, coarse clays the least.

Particle size distributions in surface (0-2 cm) sediments collected before and after the spoiling operation are presented in Table A-15. There were no major differences observed in the particle size composition of the sediments before or after the spoiling operation. The mean percentages of sand, silt, and clays in sediments collected at the Oakland Inner Harbor dredge zone were similar to sediments disposed of at the Spoil Site. After the disposal, the percentage of silts and clays in the spoiled material at the center of the Spoil Site decreased by approximately 50%. This may have been a result of removal and redistribution of the finer particles by strong bottom currents observed in this area. This possible redistribution of spoiled sediments did not appear to affect the particle size distribution of surface sediments at any of the surrounding Spoil Site stations and may have occurred only on the surface of the spoiled sediments.

Particle size distributions in settled particulates collected at Spoil Site stations before and after the experimental spoiling operation are presented in Table A-16. No significant differences were observed in particle size distribution immediately after spoiling. However, one month later consistent differences were observed in the percent of sand and fine clays in settled particulates. The mean percentage of sand increased by approximately 50% while fine clays

Table 33 Spoil Site. Surface Sediments vs. Settled Particulates. Mean percentage and one standard deviation () of sand, silt, coarse clay, and fine clay in surface sediments and settled particulates collected at Spoil Site stations. The replicate number equals four unless otherwise indicated. Resultant t ratios of a pairing design test between surface sediments and settled particulates for each sediment fraction are included. The degrees of freedom equal six.

Date:		(2 Jan./29 Jan. 1975)		(19-20 Feb./17 Mar. 1975)		t ratios
Collection:		I		II		
Group	Type					
Sand	Surface Sediment	36.9	(11)	28.2	(9)	7.36**
	Settled Particulates	6.7	(2)	14.5	(5) ^{a)}	
Silt	Surface Sediment	38.4	(6)	44.2	(5)	2.23*
	Settled Particulates	47.0	(2)	48.8	(3) ^{a)}	
Coarse Clay	Surface Sediment	7.8	(1)	9.4	(1)	8.37**
	Settled Particulates	15.2	(1)	12.4	(2) ^{a)}	
Fine Clay	Surface Sediment	16.5	(3)	18.3	(4)	8.67**
	Settled Particulates	30.6	(3)	24.2	(2) ^{a)}	

^{a)} Replicate number is five

* p<.05

** p<.01

no changes were observed in the particle size distribution of surface (0-2 cm) sediments (Table A-18).

Deposition rates and particle size distributions of settled particulates from EBMUD settling tubes are presented in Tables A-20 and A-21 respectively. The rate of deposition and the distribution of settled particulates was uniform among the four stations and through time.

A comparison of the grand means of the percentages of sand, silt, and clay in sediments and settled particulates collected at all stations during two collections is presented in Table 34. Results of a pairing design t test also included in this table indicate significant differences in the percentages of sand and coarse clay between the sediment types. However, silt and fine clay percentages were not significantly different. As was the case for Spoil Site sediments and settled particulates, the surface sediments at EBMUD stations contained higher percentages of sand than settled particulates collected at the same stations.

Table 34 EBMUD. Surface Sediments vs. Settled Particulates. Mean percentage and one standard deviation () of sand, silt, coarse clay, and fine clay in surface sediments and settled particulates collected at EBMUD stations. The replicate number equals four unless otherwise indicated. Resultant t ratios of a pairing design test between surface sediments and settled particulates for each sediment fraction are included. Degrees of freedom equal six.

Date:		(13 Dec. 1974/29 Jan. 1975)		(17 Feb./11 Mar. 1975)		t ratios
Collection:		I		II		
Group	Type					
Sand	Surface Sediment	8.8	(7.6)	7.1	(5.9)	3.12*
	Settled Particulates	.6	(.3) ^{a)}	.3	(.2)	
Silt	Surface Sediment	39.2	(2.2)	38.4	(2.2)	1.85
	Settled Particulates	43.1	(3.4) ^{a)}	38.7	(.9)	
Coarse Clay	Surface Sediment	18.4	(.4)	17.2	(2.2)	2.73*
	Settled Particulates	19.6	(1.1) ^{a)}	19.8	(.7)	
Fine Clay	Surface Sediment	35.1	(3.5)	37.3	(2.4)	1.85
	Settled Particulates	36.8	(2.8) ^{a)}	41.2	(.8)	

a) Replicate number is 4.

* $p < .05$

Similar amounts of silt were observed in surface sediments at EBMUD and the Spoil Site, whereas the Spoil Site sediments contained higher proportions of sand and EBMUD sediments contained higher amounts of clays (Tables A-15 and A-19). The composition of settled particulates was similar at both sites with slightly higher amounts of sand occurring at the Spoil Site stations (Tables A-16 and A-21). The deposition rates at the Spoil Site stations were higher than at the EBMUD stations. The estimated deposition rate at Spoil Site stations indicates that a high rate of sediment transport and redistribution occurred at this area during the period of study. No apparent relationship in the deposition rates to either the spoiling operation or proximity to the sewage outfall was observed (Tables A-17 and A-20).

2. Results of Trace Element Analyses of Benthic Sediments and Settled Particulates

a. Spoil Site

Trace element concentrations in sediments collected at four depths at the six Spoil Site stations during Collection I are presented in Table A-22. These data were subjected to a two-way analysis of variance. No significant differences between stations or depths were observed for the elements Ag, As, Cd, Cr, Cu, and Pb (Table 35).

A highly significant difference in Hg concentrations between depths was observed; the highest concentrations occurred in the 10-30 and 30-60 cm layers and the lowest concentrations were in the 0-10 cm depth. Significant differences between stations were found during Collection I in the concentrations of the elements Fe, Hg, Mn, Ni, Se, and Zn. In general, these elements were in the highest concentrations at station 6 and in the lowest concentration at station 1.

Trace element concentrations in surface (0-2 cm) sediments collected at the six Spoil Site stations before and after the spoiling operations are presented in Table A-23. The concentrations of the elements Ag, As, Cd, and Cr were not significantly different among stations or collections (Table 36). Significant differences were observed in Cr, Fe, Mn, and Zn concentrations between stations with the highest concentrations occurring at station 6 and the lowest at station 1. Although no significant differences among stations or collections were observed in Hg and Ni concentrations, significant interaction between stations and collections occurred. Significant differences in the elements Cr, Fe, Mn, Pb, Se, and Zn were found between collections.

Group pairing design t tests, presented in Table 37, indicate that the increased concentrations of Cu and Fe observed at the inner stations were related to the spoiling operation. The other elements did not increase or decrease in relation to the spoiling operation.

Trace element concentrations in settled particulates collected at the six Spoil Site stations during three collections are presented in Table A-24. No significant differences in the concentrations of the elements were observed either between stations or between collections (Table 43).

The data for the collections immediately before and after the spoiling operation were subjected to a group pairing design t test (Table 38). Changes observed at inner stations were not significantly different from changes observed at outer stations during the spoiling operation.

The grand mean concentrations of ten elements in surface sediments and settled particulates are presented in Table 39. The data were subjected to a pairing design t test to determine whether the settled particulates contained the same trace element concentrations as adjacent surface sediments in the Spoil Site area. Significantly higher concentrations of the elements Cu, Fe, Hg, Mn, Ni, Pb, and Zn were observed in settled particulates. Differences in the

Table 35 Spoil Site. Sediments. Stations vs. Sediment Depth. Results of two-way analysis of variance of metal concentrations in four sediment layers (0-2, 2-10, 10-30, and 30-60 cm) at the six Spoil Site stations during Collection I (26 Dec. 1974 - 2 Jan. 1975). Degrees of freedom equal (df).

Groups	(df)	Ag	As	Cd	Cr	Cu	Elements							Zn
							Fe	Hg	Mn	Ni	Pb	Se		
Depths	(3)	1.97	1.69	.11	.55	2.82	1.77	15.9***	1.50	1.89	1.32	1.05	1.93	
Stations	(5)	1.51	.75	.61	.81	2.19	3.56*	7.35***	5.26**	4.34*	2.92	3.50*	4.36*	
Depth X Sta.	(15)	1.76	1.10	2.05	.75	.91	.73	1.12	.65	.62	.95	.70	.47	
Within subgroups (48)														

* p<.05
 ** p<.01
 *** p<.001

Table 36 Spoil Site. Surface sediments. Resultant F ratios from a two-way analysis of variance of element concentrations in surface sediments collected at six Spoil Site stations during Collections I (26 Dec. 1974-2 Jan. 1975), II (29 Jan. 1975), and III (19-20 Feb. 1975). Degrees of freedom equal (df).

Groups	(df)	Ag	As	Cd	Cr	Cu	Elements				Ni	Pb	Se	Zn
							Fe	Hg	Mn					
Stations	(5)	.95	1.10	.71	3.38*	1.51	7.05***	.80	4.26**	.65	1.76	1.77	4.20**	
Collections	(2)	1.42	.36	1.13	5.31**	2.75	11.0***	.01	13.3***	.17	5.40**	4.18*	11.8***	
Sta. x Coll.	(10)	.98	1.20	2.07	1.13	1.13	1.61	5.62*	2.07	4.06*	1.22	1.83	1.61	
Within subgroups	36													

* p<.05
 ** p<.01
 *** p<.001

Table 37 Spoil Site, Surface Sediments (0-2 cm). Group Pairing Design Test. A comparison of changes in mean element concentrations observed at stations outside the spoil area before and after spoiling operations and at stations inside of the spoil area during this same period. \bar{X}_{d_0} = mean difference (change) at outside stations 1, 3 & 5; \bar{X}_{d_i} = mean difference at inside stations 2, 4 & 6. Degrees of freedom equal four.

Element	t	\bar{X}_{d_0}	\bar{X}_{d_i}	$(\bar{X}_{d_i} - \bar{X}_{d_0})$
Ag	<1	.20	1.0	.80
As	1.20	3.13	-.57	-3.70
Cd	1.95	-.13	2.43	2.57
Cr	<1	-24.00	-34.00	-10.0
Cu	6.80*	-1.33	-13.00	-11.67
Fe %	3.79*	-.29	-.78	-.49
Hg	<1	.029	-.017	-.047
Mn	<1	-67.67	-109	-41.33
Ni	<1	-4.33	10.00	14.33
Pb	<1	-9.70	-15.10	-5.40
Se	<1	-1.43	-1.07	.37
Zn	4.00	-10.33	-31.67	-21.33

* p<.05

particle size distribution of surface sediments and settled particulates could account for the observed differences in element concentrations between surface sediments and settled particulates. However, Serne and Mercer (1975) concluded that most of the variation in element concentrations in San Francisco Bay sediments was related to the sulfide content rather than to particle size of sediments.

b. EBMUD

The mean concentrations of the twelve trace elements in sediments collected from four depths at the EBMUD stations during Collection I are presented in Table A-25. A two-way analysis of variance indicates significant differences in the concentrations of the elements As, Cd, Fe, and Ni between depths (Table 40). Cadmium concentrations were highest in the deeper layers and lowest at the

Table 38 Spoil Site. Settled Particulates. Group Pairing Design Test. A comparison of changes in mean element concentrations observed at stations outside the spoil area before and after spoiling operations and at stations within the spoil area during this same period. \bar{X}_d = mean difference (change) at outside stations 1, 3, and 5; \bar{X}_i = mean difference at inside stations 2, 4, and 6. Degrees of freedom equal three. a)

Element	t*	\bar{X}_d	\bar{X}_i	$(\bar{X}_i - \bar{X}_d)$
Ag	<1	-1.5	.3	.8
As	<1	.8	-1.2	-1.0
Cd	<1	.3	1.0	.7
Cr	<1	21.5	-45.1	-66.6
Cu	1.36	-4.8	2.4	7.2
Fe	<1	-1158.2	-98.5	-1459.7
Hg	<1	-.30	-.05	.25
Mn	1.14	246.7	-334.9	-581.5
Ni	1.28	187.1	-106.3	-273.4
Pb	<1	-6.3	-2.9	3.4
Se	<1	-1.0	-1.2	-.2
Zn	<1	-6.8	-7.4	-.5

* significant difference at $(p = .05) = 3.18$.

a) Settling tubes missing at some stations and some collection periods.

surface while the reverse was true for As, Fe, and Ni concentrations. Nickel concentrations were significantly higher at Station 1. Significant interaction between stations and depths was observed for the elements Ag, Cr, Cu, Hg, and Mn.

Trace element concentrations in surface sediments collected at the EBMUD stations during two collections are presented in Table A-26. No significant differences were observed between collections or stations in Ag, Cd, Cr, Pb, or Se concentrations (Table 41).

Arsenic, Cu, Fe, Ni and Zn concentrations were significantly different between stations. Concentrations of As, Cd, Cu, Fe, Ni and Zn were highest at the stations nearest the EBMUD outfall and may have resulted from the local input of these metals from this source. Arsenic concentrations decreased significantly between the two collections and significant interaction between collections and

Table 39 Spoil Site. Sediments and Settled Particulates. Results of a pairing design t test of the differences in ten elements between surface sediments and settled particulates collected at all stations during all collections. Concentrations are in parts per million (except Fe). Degrees of freedom equal 10.

Element	Surface Sediments	Settled Particulates	t
As	8	11	1.87
Cr	189	235	2.22
Cu	44	67	3.91**
Fe (%)	3.4	4.8	3.54**
Hg	.27	.43	5.63**
Mn	458	770	3.99**
Ni	120	236	2.98*
Pb	36	53	3.28**
Se	2.1	2.3	.43
Zn	112	156	3.76**

* $p < .05$

** $p < .01$

stations was observed in Hg and Mn concentrations.

The mean concentrations of the twelve elements in settled particulates collected at the four EBMUD stations during three collections are presented in Table A-27. Significant differences between stations were observed only in the concentrations of Hg and Ni (Table 42). Mercury concentrations decreased continuously through time while Ni concentrations increased sharply prior to Collection II, then decreased to levels comparable to those observed during Collection I.

Grand mean concentrations of ten elements in surface sediments and settled particulates collected at EBMUD stations during all collections, are compared in Table 44. The pairing design t ratios also presented in Table 44 indicate significantly greater concentrations of the metals Cu, Fe, Hg, and Zn in settled particulates than in surface sediments. As in the case of the Spoil Site sediments, the higher concentrations of these metals in settled particulates at the EBMUD site also appeared to be related to the higher percentages of silt and clays in these samples.

Table 40 EBMUD Sediments. Stations vs Sediment Depth. Results of two-way analysis of variance of metal concentrations in four sediment layers (0-2, 2-10, 10-30, 30-60 cm) at the four EBMUD stations during Collection I (13-14 Dec. 1974). Degrees of freedom equal (df).

Element:	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Groups (df)												
Depths (3)	.41	3.84*	3.29*	.26	.28	3.80***	.43	1.79	4.55**	1.20	1.40	.65
Stations (3)	.29	1.04	1.51	1.15	.47	1.50	.38	.32	14.42***	.38	.24	2.13
Depth x Sta. (9)	3.19*	1.37	2.04	2.86*	2.60*	.94	12.28***	14.75***	1.23	1.57	1.92	1.88
within subgroups (32)												

* p<.05
 ** p<.01
 *** p<.001

Table 41 EBMUD Surface Sediments. Stations vs. Time. Results of two-way analysis of variance of metal concentrations in surface sediments collected at the four EBMUD stations during Collection I (13 Dec. 1974) and Collection II (17 Feb. 1975). Degrees of freedom equal (df).

Element:	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Groups (df)												
Stations (3)	.75	6.77**	1.99	1.77	3.50*	11.37***	.69	8.73	131.59***	2.30	2.29	4.32
Collections (1)	2.22	7.05*	2.07	1.50	.01	.80	5.07	3.11	.55	.14	.84	2.03
Sta. x Coll. (3)	.25	1.01	.73	1.25	.63	.33	4.00*	3.95*	1.15	.75	1.44	.36
within subgroups (16)												

* p<.05
 ** p<.01
 *** p<.001

Table 42 EBMUD. Settled Particulates. Resultant F ratios of a two-way analysis of variance without replication of element concentrations in settled particulates collected at the four EBMUD stations during Collections I (10 Jan. 1975), II (11 Feb. 1975), and III (18 Mar. 1975). Degrees of freedom equal (df).

Group	(df)	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Stations	(2)	-	.28	-	.34	.58	.11	16.22*	4.10	10.37*	1.40	.50	.08
Collections	(2)	-	4.20	-	.69	.01	.29	1.03	.47	.68	.15	.50	.21
Sta. x Coll.	(4)												

* p<.05

Table 43 Spoil Site. Settled Particulates. Resultant F ratios from a two-way analysis of variance without replication of element concentrations in settled particulates collected at Spoil Site during Collections I (29 Jan. 1975), II (8 Feb. 1975), and III (17 Mar. 1975). Degrees of freedom equal (df).

Group	(df)	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Stations	(4)	-	.77	-	.35	.33	.40	.36	.32	.08	.24	1.24	.45
Collections	(2)	-	3.92	-	.46	1.00	1.24	.44	.02	.35	.41	.32	1.32
REPE	(8)												

no significant differences observed.

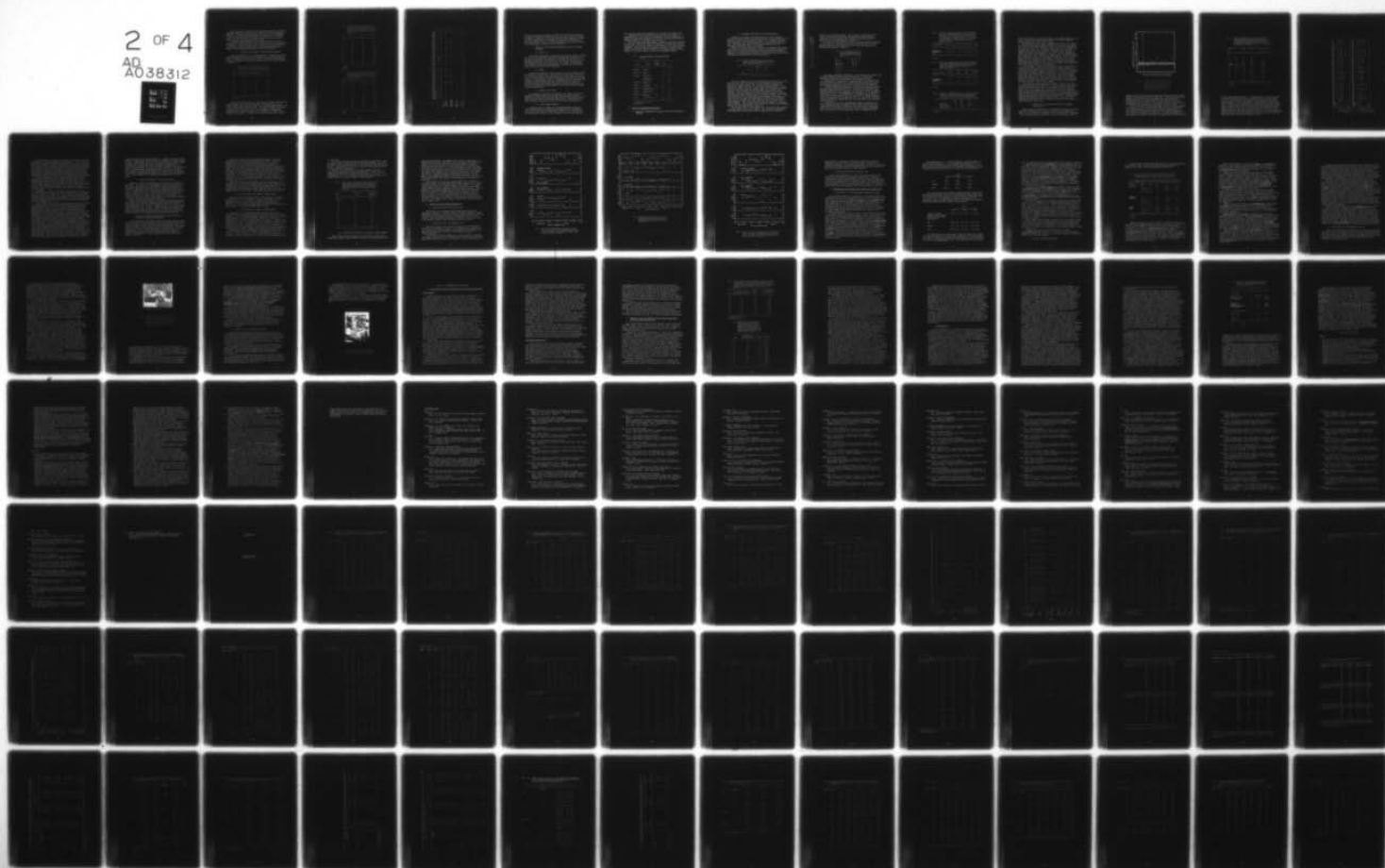
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The grand mean concentrations of the twelve elements in surface sediments collected at Spoil Site and EBMUD stations are compared in Table 45. Significantly higher concentrations of all the elements except Ag, Cd, and Se were observed at the EBMUD site. The higher levels of these elements at the EBMUD stations may reflect past or present inputs from the EBMUD sewage outfall.

Grand mean concentrations of the twelve elements in settled particulates collected at Spoil Site and EBMUD stations are presented in Table 46. Concentrations of the elements Cu, Hg, Pb, and Zn were significantly higher in EBMUD samples. Selenium concentrations, however, were significantly higher at Spoil Site stations.

c. Oakland Inner Harbor - Dredge hopper sediments

Mean concentrations of the trace elements in sediments collected from the dredge site prior to the dredging, from the dredge hoppers prior to disposal, from the center of Spoil Site after disposal, and from the six Spoil Site stations during Collection III are presented in Table 47.

Table 44 EBMUD. Sediments and Settled Particulates. Results of a pairing design t test of the differences in ten elements between surface sediments and settled particulates collected at all stations and all collections. Concentrations are in parts per million (except Fe). Degrees of freedom equal 6.

Element	Surface Sediments	Settled Particulates	t
As	13	12	.28
Cr	246	241	.54
Cu	70	81	3.18*
Fe (%)	3.9	4.6	2.63*
Hg	.48	.72	8.87***
Mn	678	780	1.23
Ni	174	223	1.34
Pb	65	72	2.30
Se	2.0	2.7	2.00
Zn	168	191	3.39*

* $p < .05$

*** $p < .001$

These data provide a means of assessing the transfer and input of trace elements into the Spoil Site zone. Concentrations of the twelve elements were generally higher in the dredge hopper samples than at any of the other three locations. The lowest concentrations were generally found in the Spoil Site station samples, although the concentrations of several elements were comparable to dredge zone (Oakland Inner Harbor) or impact zone (center of Spoil Site) samples.

Table 45 Surface Sediments. Spoil Site vs. EBMUD. Results of a pairing design t test of the differences between the grand mean concentrations in parts per million dry weight of twelve elements in surface sediments collected at Spoil Site stations during Collections I and III and at EBMUD stations during Collection I and II. Degrees of freedom equal 18.

Element	Spoil Site	EBMUD	t
Ag	.8	.9	.39
As	8.6	12.6	2.22*
Cd	.6	.5	.23
Cr	205	246	3.43**
Cu	45.4	70.3	8.94***
Fe	34,543	43,824	4.67**
Hg	.27	.48	7.44***
Mn	470	678	3.64**
Ni	115	173	2.91**
Pb	36.6	64.7	5.37**
Se	2.2	1.6	1.08
Zn	116	168	6.98**

* p<.05
 ** p<.01
 *** p<.001

Table 46 Settled Particulates. Spoil Site vs. EBMUD. Results of a pairing design t test of the differences between the grand mean concentrations in parts per million dry weight of twelve elements in settled particulates collected at Spoil Site and EBMUD stations during three collections. Degrees of freedom equal 24.

Element	Spoil Site	EBMUD	t
Ag	.8	.6	.42
As	11.2	12.4	.97
Cd	.9	.3	1.99
Cr	250	241	.41
Cu	66.3	81.0	2.36*
Fe _s	4.65	5.19	1.17
Hg	.48	.65	2.53*
Mn	812	745	.68
Ni	241	208	.74
Pb	55.8	69.8	2.80**
Se	2.7	2.0	2.08*
Zn	158	189	2.28*

* p<.05
 ** p<.01

Table 47 Sediments. Mean concentrations in parts per million (except Fe) and one standard deviation () of twelve elements in sediments collected from Oakland Inner Harbor prior to dredging, from the hoppers of the dredge ship CHESTER HARDING just prior to spoiling, from the center of the Spoil Site after spoiling, and from the six Spoil Site stations during Collection III. The replicate number (n) is included.

Location	n	Elements											
		Ag	As	Cd	Cr	Cu	Fe%	Hg	Mn	Ni	Pb	Se	Zn
Oakland Inner Harbor (30 Jan 1975)	3	<2	5(4)	<2	301(127)	70(28)	3.5(1.2)	1.22(.43)	459(164)	117(44)	78(36)	1(.2)	168(74)
CHESTER HARDING dredge hopper (3-4 Feb 1975)	21	<2	15(7)	<2	319(52)	124(34)	5.0(1.1)	2.46(.51)	710(227)	260(135)	117(26)	3(1)	286(66)
Spoil Site Impact Zone (20 Feb 1975)	1	<2	12	<2	278	51	3.0	1.7	349	117	48	.4	139
Spoil Site Stations 1-6 (20 Feb 1975)	18	<2	8(3)	<2	212(15)	47(5)	3.7(.3)	.27(.05)	521(46)	126(33)	44(6)	3(1)	126(13)

The particle size data presented in Table A-15 indicate differences in the size distribution of sand, silt and clays among these four groups of samples. Comparison of the above data and data presented in Table A-15 also suggest that trace element concentrations increase with decreasing particle sizes. A further discussion of this relationship is presented below (Integration and Discussion).

3. Results of Chlorinated Hydrocarbon Analyses of Surface Sediments

a. Spoil Site

Chlorinated hydrocarbon concentrations in surface sediment samples from the two pre-spoiling collections are presented in Table A-28. Significant decreases were observed in the mean concentrations of PCB and DDE between these two collections. The average concentration of PCB decreased almost 30% from 26.1 ppb to 18.5 ppb. No differences were observed in the mean concentrations of chlorinated hydrocarbons between stations during either collection.

b. EBMUD

Chlorinated hydrocarbon concentrations at the four EBMUD stations are presented in Table A-29. During the first collection, mean PCB and DDE concentrations were approximately 3-4 times higher at stations 1 and 2 than at stations 3 and 4. Two months later this trend was not observed. Storm-related increases in freshwater input and street runoff which occurred between the two collections did not appear to increase the total chlorinated hydrocarbon sediment burden since the mean of concentrations of PCB and DDE at all stations were equivalent in the two collections.

c. Oakland Inner Harbor

The sediment collected at the Oakland Inner Harbor just prior to dredging contained a mean PCB concentration of 59 ppb, which was slightly more than three times the mean concentration observed at the Spoil Site area during this period (Table A-30). Redistribution of these sediments constituted a source of PCB input into Spoil Site.

d. Dredge hopper sediment

Sediments from the hoppers of the CHESTER HARDING contained higher concentrations of chlorinated hydrocarbons than sediments in any of the three areas discussed above (Table A-30). The mean PCB concentration of 130 ppb in these sediments was seven times greater

than the PCB concentrations observed at the Spoil Site (Table A-28).

Concentrations of p,p'-DDE and p,p'-DDD were also higher in dredged material than in Spoil Site sediments. The concentrations of these DDT analogs were generally low and p,p'-DDT was not detected in sediments except at a few Spoil Site stations.

Data for chlorinated hydrocarbons in sediments from San Francisco Bay and other areas of the world are presented in Table 48 as a means of assessing the relative degree of contamination of San Francisco Bay sediments. San Francisco Bay sediments contained chlorinated hydrocarbons concentrations which are within the range of reported values; several of the other reported values are considerably higher.

Table 48 Chlorinated hydrocarbons in sediments from global areas. Values are expressed as ppb dry weight unless otherwise indicated.

Reference	Area Examined	Approximate date measured	Total DDT a)	PCB
Nimmo <i>et al.</i> , 1971	Escambia Bay, Florida		NM	0.6-61.0 b)
Dexter and Pavlou, 1973	Saronikos Gulf, Greece		7.1-1893	1.3-775
Young <i>et al.</i> , 1973	Los Angeles Co. Sewage Outfall		26 b)	0.1-10 b)
	Orange Co., Calif. Sewage Outfall		70	30-90
Halcrow <i>et al.</i> , 1974	Firth of Clyde, Scotland		NM	10-2890
Hom <i>et al.</i> , 1974	Santa Barbara Basin, Calif.	1965	139.3	103
Law and Goerlitz, 1974	San Francisco Bay Tributaries, Calif.	1972	4-373	1.2-1400
Linko <i>et al.</i> , 1974	Turku Archipelago, Finland		NM	10-20 c)
Hom and Pavlou, 1975	Puget Sound, Washington	1974	20.3	609
This study	San Francisco Bay	1975	2-20	20-120

a) Total DDT includes DDD and DDE.

b) Parts per million

c) Wet weight

D. Water and Suspended Particulates

1. Results of Suspended Particulate Load and Trace Element Analyses

a. Suspended particulate load determinations

Suspended particulate load concentrations in water samples collected at the six Spoil Site stations before, during, and after the spoiling operation are presented in Table A-31. Sediment load data for Collections I and II were subjected to a group pairing design t test. No significant effect of the spoiling operation on the suspended particulate loads was observed.

Results of one-way analysis of variance of suspended particulate loads between stations during each collection period are presented in Table 49. No significant difference was observed between stations; average station particulate loads for each collection period were used therefore as replicates for all further analyses.

Table 49. Spoil Site. Suspended Particulates. Resultant F ratios of one-way analyses of variance in particulate load between stations for top and bottom samples during each collection period. Degrees of freedom equal (df).

	Collections					
	df	I	F	df	II	F
Top	8	<1	8	<1	-	-
Bottom	8	<1	8	<1	8	<1

Since top samples were not collected during Collection III, only data from the first two collections were subjected to three-way analysis of variance (Table 50). No significant differences in particulate loads among the size fractions, between depths, or with time were observed. However, highly significant interaction between particle size and depths and between depth and collection were observed. Coarse particle sizes were most abundant and the fine particle sizes least abundant in bottom samples, whereas the medium sized particles were most abundant in top samples (Table A-33). This would account for the highly significant interaction term between particle size and depth and would mask any differences in particle loads among the middle and coarse particle sizes in either top or bottom samples. This interaction was examined further in a multiple range analysis of particulate loads within each particle size fraction between top and bottom samples (Table 51).

Particulate loads of coarse particles were significantly greater in bottom samples than in top samples. No significant differences were observed in particulate loads of medium and fine particles

between top and bottom samples. The difference in the particulate load of coarse particles between top and bottom samples during collection I and II explains the significant interaction between particle size and depth observed in Table 50.

To resolve the significant interaction observed between depth and collection period, the total particulate loads observed in top and bottom samples during collection I and II were also subjected to a multiple range analysis (Table 52).

Table 50. Spoil Site. Suspended Particulates. Resultant F ratios of a three-way analysis of variance, model II (with replication) of particulate loads in three particle size fractions, from two depths, between collections I and II. Degrees of freedom equal (df).

	df	EMS	F	ddf
Particle size	2	50.14	10.48	1.55
depth	1	14.85	1.29	3.32
Coll. I & II	1	0.00	<1	----
Part. size X depth	2	5.309	9.27***	
Part. size X coll.	2	.54	<1	
Depth X coll.	1	7.30	12.75***	
Part. size X depth X coll.	2	1.07	1.86	
Within subgroups	36	.573		

*** p<.001

The highly significant difference in particulate loads between top and bottom samples observed during Collection II explains the significant interaction between depth and collection period.

Differences in the particulate loads between the three particle sizes are obvious in Table A-31. These data were subjected to a two-way analysis of variance of the particle load within each particle size in bottom samples from all three collection periods (Table 53).

A highly significant difference in particulate loads was observed among the three size fractions, but no significant differences were observed between collection periods. The particulate load of coarse, medium and fine particles appeared to be equivalent in top samples in Collections I and II. The differences in sediment load ratios of the three particle sizes between top and bottom and the changes through time in top samples would explain the lack of a significant difference in particulate loads between particle sizes observed in the three-way anova (Table 50).

The variation of total suspended particulates between top and bottom samples through time resulted mainly from the fluctuations of coarse particulate loads. Medium and fine particulates settle more slowly than do coarse particulates and are more readily resuspended. This could account for their relatively stable concentrations and

Table 51. Spoil Site. Suspended Particulates. Tukey's w procedure of multiple range analysis of mean particulate loads of three particle size ranges between top and bottom samples from collections I and II. Means not overlapped by the same solid line are significantly different. T = top samples, B = bottom samples, F = particle sizes 1 μ to .07 μ , M = particle sizes 4 μ to 1 μ , C = particle sizes >4 μ , df = degrees of freedom.

Size x Depth	TF	BF	TC	TM	BM	BC
	**					
<u>\bar{x} suspended</u> particulate load mg/l	1.94	2.41	4.30	4.65	5.07	6.74

** p<.01

Table 52. Spoil Site. Suspended Particulates. Tukey's w procedure of multiple range analysis of mean particulate loads for all top and all bottom samples from collections I and II. Means not overlapped by the same solid line are significantly different.

Depth x Collection	Coll. II, Top	Coll. I, Top	Coll. I, Bot.	Coll. II, Bot.
	**			
<u>\bar{x} suspended</u> particulate load mg/l	3.25	4.01	4.34	5.14

** p<.01

Table 53. Spoil Site. Suspended Particulates. F ratios of a two-way analysis of variance, model II, of particulate loads in three particle size fractions in bottom samples from collection I, II and III. Degrees of freedom equal (df).

	(df)	EMS	F
Particle size	2	26.04	37.53***
Collections	2	1.71	2.47
Part. size x coll.	4	.63	<1
Within subgroups	9	.69	

***p<.001

uniform distribution in the water column. The changes observed at the center of Spoil Site indicate that almost all of the differences between Collections I and II were natural.

Suspended particulate loads in samples collected at the center of the Spoil Site during the disposal operation are presented in Table A-32. Each of the seven disposals occurred at or near a maximum ebb or flood tide (Table 2). A 24-liter water sample was collected five minutes prior to the first disposal. Subsamples for a 24-liter composite sample were collected within the spoil plume immediately after each of the seven disposals. Composite 24-liter samples were also collected 1.5 and 3 hours after each disposal. These two samples were collected at slack and ebb periods of the tidal cycle. Subsamples for the 24-liter 4.5 hour composite sample were collected during the first maximum flood or ebb tide following each disposal. A sixth sample was collected at the center of the Spoil Site, during an ebb tide, 9 hours after the last disposal.

The grand mean, range, and 95% confidence limit of particulate loads in all samples collected at Spoil Site stations during all collections were compared with the particulate load data from the six time sequence composite samples collected at the center of the Spoil Site during the disposal operation (Figure 5). Particulate loads within the spoil plumes were 23 times greater than the grand mean particulate loads of all samples collected at Spoil Site stations. During the daylight hours, the spoil plume was visible for less than 20 minutes and the high particulate loads observed in plume samples were not observed in the 1.5 hour samples.

The particulate loads observed in samples collected 4.5 hours after each disposal and 9 hours after the last disposal were twice as high as particulate loads observed in samples collected 5 minutes before the first disposal and 1.5 and 3 hours after each disposal. The 4.5 and 9 hour samples were outside the range of the Spoil Site station grand particulate load mean, however, the average particulate load of the five center samples (excluding spoil plume samples) is not significantly different from the station grand means. This suggests that the increased particulate loads observed in 4.5 and 9 hour samples were probably a result of natural particulate load fluctuations during the tidal cycle.

b. Results of trace element analyses of suspended particulates

Trace element concentrations of suspended particulates collected at Spoil Site stations during three collections are presented in Table A-33. Differences in element concentrations between Collection I and II (before and during spoiling) in all three particle sizes in

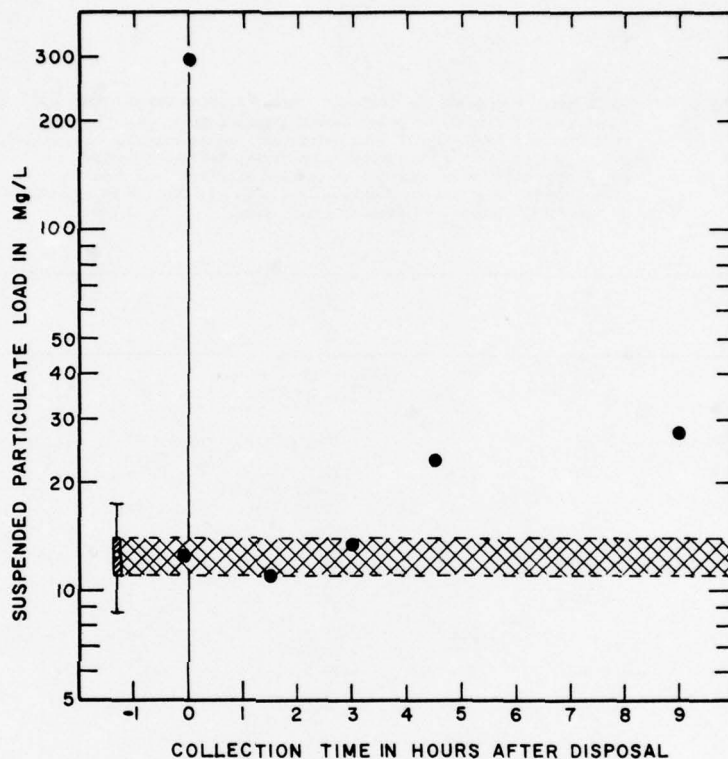


Figure 5. Comparison of suspended particulate load in water at center of Spoil Site at various times with respect to disposal times (filled circle points) with the load at all Spoil Site stations for all collection times. Crosshatched area - the 95% confidence limits for the mean total load of all stations for all collections. The range of the individual values is also shown.

samples collected from inner and outer Spoil Site stations were subjected to a group pairing t -test (Table 54). Significant changes were observed only in Cu and Fe concentrations during this period. Concentrations of Fe increased more at outer stations after the experimental disposal than at inner stations, while Cu concentrations increased at outer stations and decreased at inner stations.

The data in Table A-33 were subjected twice to a two-way analysis of variance since only bottom samples were collected in Collection III. Two-way analysis of variance of element concentrations in three particle size fractions collected at two depths during Collection I and II are presented in Table 55. Significant differences were observed in element concentrations between particle size fractions

Table 54 Spoil Site, Suspended Particulates. Group Pairing Design Test. A comparison of changes in mean element concentrations observed at stations outside the spoil area before and during spoiling operations, and at stations within the spoil area during this same period. $\bar{X}d_0$ = mean difference (change) at outside stations 1, 3 & 5; $\bar{X}d_1$ = mean difference at inside stations 2, 4 & 6. Values for Fe are expressed as percents. Degrees of freedom equal four.

Element	t	$\bar{X}d_0$	$\bar{X}d_1$	($\bar{X}d_1 - \bar{X}d_0$)
Ag	<1	5.9	6.6	.7
As	2.18	-2.7	4.1	6.7
Cd	1.511	2.9	12.2	9.2
Cr	1.5	-27.4	13.8	41.2
Cu Top	<1	62.3	180.3	118.0
Cu Bottom	2.99*	-58.0	20.0	78.0
Fe	4.45**	-4.2	-.1	4.1
Hg	<1	0.17	0.12	-0.05
Mn	1.32	2.4	-129.7	-132.1
Ni	<1	-5.7	-39.5	-33.8
Pb	<1	-40.5	-21.1	19.4
Se	<1	.7	.5	-.2
Zn	<1	-38.5	-25.2	13.3

* $p < .05$

** $p < .01$

in all elements except As and Se. Except for Mn, the highest element concentrations were observed in the smaller particle sizes (Table A-33); Mn concentrations were highest in the medium-sized particles. Crecelius et al. (1975) observed that concentrations of Hg and total organic carbons in bottom sediments increased with decreasing particle size. In the present study, a similar relationship may have existed between organic carbon, particle size and all the elements examined except As and Se. The high ion exchange capacity of the fine clay particles may also have further contributed to the inverse relationship observed between total trace element concentrations and particle size (Pearson et al., 1970; Serne and Mercer, 1975).

Table 55 Suspended Particulates. Results of a two-way analysis of variance in the mean concentrations of twelve elements in three particle size fractions collected at all Spoil Site stations at two depths during collections I and II. Degrees of freedom (df) are included.

Element	(df)	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Particle size	2	17.59***	2.17	6.36**	14.24***	84.95***	34.36***	17.44***	35.82***	3.60*	75.92***	1.28	15.73***
Depth	1	<1	<1	<1	<1	19.28***	<1	<1	<1	<1	<1	<1	2.68
Part. size x depth	2	<1	1.15	<1	<1	<1	1.42	<1	<1	1.99	1.07	<1	<1
With subgroups	30												

Table 56 Suspended Particulates. Results of a two-way analysis of variance in the mean concentrations of twelve elements in three particle size fractions collected at all Spoil Site stations at bottom depths during all three collections. Degrees of freedom (df) are included.

Element	(df)	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Particle size	2	15.28***	<1	3.08	7.40*	22.95***	39.86***	6.55*	39.97***	3.61	12.96*	2.27	43.17***
Collection	2	7.13**	<1	1.12	5.75*	<1	16.52***	2.08	16.63**	4.28	3.45	<1	33.48***
Part. size x coll.	4	1.85	<1	11.44**	1.50	<1	<1	<1	<1	6.88**	4.15*	<1	3.05
Within subgroups	9												

* P<.05

** P<.01

*** P<.001

Two-way analysis of variance of element concentrations in three particle sizes in bottom samples from Collections I, II and III are presented in Table 56. Significant differences in element concentration between particle sizes were observed in all elements except As, Cd, Ni, and Se. The differences in Cd and Ni concentrations observed between particle sizes in the first two collections (Table 54) were not indicated when only bottom samples from all three collections were tested. A significant interaction between particle size and collection was observed which indicates that element concentrations fluctuated among the three particle sizes during the three collections.

Only Cu concentrations were significantly different between depths (Table 55) and Ag, Cr, Fe, Mn and Zn concentrations were significantly different between collections (Table 56). Mean Cu concentrations were higher in top samples during Collection I and II than in bottom samples. This difference and the observed differences in particulate loads between particle size fractions may have partially resulted from the greater concentrations of phytoplankton in the top samples.

Significant differences in element concentrations in bottom sample particulates between collections were observed in the elements Ag, Cr, Fe, Mn and Zn (Table 56). Iron, Mn and Zn concentrations increased prior to Collection II (during spoiling) and then decreased to levels below pre-spoiling concentrations. Chromium concentrations increased gradually through time while Ag levels decreased markedly prior to Collection III.

Trace element concentrations in suspended particulates collected at the impact center of the Spoil Site during spoiling operations are presented in Table A-34. The highest concentration of the elements Ag, Cd, Cu, Fe, Pb, Se, and Zn were associated with the smallest sized particulates while Cr and Ni concentrations were equivalent in all three particle sizes. Arsenic concentrations were equivalent in coarse and fine sized particles and lowest in medium sized particles. High Hg concentrations were observed in each of the size classes depending on collection period, but were generally higher in medium and fine particulates. Only Mn concentrations were consistently lower in fine particulates and equivalent in medium and coarse particles. The concentrations of the elements Ag, As, Cd, Cu, Fe, Mn, Ni, Pb, Se, and Zn decreased in samples collected within the spoil plume immediately after each disposal. Concentrations of Ag, Cd, Cu, Mn and Ni decreased by over 50% from levels observed in particulates collected 5 minutes before the first disposal while only slight decreases were observed in As, Pb, Se, and Zn concentrations. Only Cr and Hg concentrations increased immediately after each of the seven disposals. Mercury concentrations in dredge hopper sediments were three times higher than in suspended particulates collected

immediately before the spoiling operation. Chromium levels were also higher in dredge hopper sediments than in suspended particulates before spoiling. Although As, Fe, Pb, and Zn concentrations in dredge hopper sediments were slightly higher than pre-spoiling particulate concentrations, particulate levels decreased in samples collected within the spoil plume. The cause of these decreases are not presently known. Although lower concentrations of Ag, Cd, Cr, Mn, Ni and Se were observed in dredge hopper sediments in relation to pre-spoiling particulates, dredge hopper sediments could have diluted the concentrations of these elements in particulates collected within the spoil plume.

c. Results of analyses of metals in solution

Cadmium, Cu, and Pb concentrations in solution are presented in Table A-35. The results of Hg determinations are not included since the samples were found to be contaminated with Hg, probably during preparation (Bothner and Robinson, 1975). Concentrations of Pb, Cd, and Cu in samples collected in the disposal plume were respectively 9, 6, and 4 times greater than concentrations observed in other samples collected during this study. Lead, Cd, and Cu concentrations in samples collected at stations and at the center of the Spoil Site 3 hours after the sixth disposal were similar to concentrations observed in samples collected one month after spoiling.

Metal concentrations observed in the present study were consistently lower, but in general agreement with concentrations observed in San Francisco Bay waters by Serne and Mercer (1975). During the disposal operation, an increase in soluble Pb, Cd, and Cu was observed which lasted less than 1.5 hours. These increases were greater than those observed by Serne and Mercer (1975) in their batch desorption studies and were probably eliminated either by mixing and dilution by tidal currents or by re-adsorption onto suspended particulates.

2. Results of Chlorinated Hydrocarbon Analyses of Water, Including Suspended Particulates

In the laboratory, each sample column was extracted by successive 500 ml volumes of acetone and hexane which were passed into a 2 liter separatory funnel to which 800 ml of distilled water were added. The funnel was shaken, and the water-acetone phase discarded. The hexane phases of these water column extracts were concentrated to volumes of approximately 1 to 2 ml using modified micro-Snyder columns in a thermostatically controlled heating apparatus (Kontes Glass Co.). Samples were analyzed on the gas chromatograph system described in Part II, Section C.

Chlorinated hydrocarbon concentrations in water, including suspended particulates, are presented in Table A-36. Except for marked increases in chlorinated hydrocarbon levels during and immediately following several of the experimental disposals, chlorinated hydrocarbon concentrations remained stable throughout the study. The PCB and DDE levels in January and March were equivalent, with PCB concentrations approximately an order of magnitude higher than those of either DDE or DDD. Significant increases in the levels of chlorinated hydrocarbons were seen only during the spoiling operations. Concentrations of PCB in sample #8, collected at the center of the Spoil Site immediately after the fourth disposal, were more than ten times greater than sample #9 collected at the same site one-half hour later.

Many unknown peaks were observed in the chromatograms of water samples collected during the spoiling operation. These peaks, which survived both sulfuric acid cleanup and on-column saponification, did not match the profiles of any of the lower-chlorinated PCB's or commonly observed pesticides, and had retention times which were much shorter than that of p,p'-DDE. These compounds were not identified.

3. Results of Analyses of Composite Sewage Samples

Water samples were not collected in the vicinity of the EBMUD outfall owing to time limitations. To gain information concerning pollutant input into the outfall vicinity and to compare the magnitude of these inputs with the dredge spoiling operations, a number of one-week composite sewage effluent samples were analyzed for trace element and chlorinated hydrocarbon content.

a. Trace elements in composite sewage samples

Twenty-four-hour, flow-proportional composite samples were collected daily by EBMUD personnel using a Chicago "True Test" water sampler. A flow proportional composite sample is a series of constant volume samples (25 ml each) taken over variable time intervals depending upon the volume of water passing the device. These effluent samples were assumed to be representative of material discharged into the receiving waters. Each day, for 6 to 7 consecutive days, a 150 ml aliquot of the 24 hour composite was composited in a refrigerated acid-leached polyethylene bottle. These samples covered the period of time for which transplant animals were exposed at the EBMUD stations.

These samples were centrifuged in acid-leached polyethylene bottles to remove particles $>0.07\mu$ esd (estimated sp. gr.: 2.65) or $>0.4\mu$ esd (estimated sp. gr.: 1.07). The collected sediment was prepared for X-ray fluorescence and IZAA analyses as described in Part

II, Section B.

The results of these analyses are presented in Table A-37. The particulate loads remained relatively constant throughout the period of study as did the concentrations of the elements As, Fe, Hg, Mn, Ni, and Se. Concentrations of Ag, As, Cd, and Se varied between collections by an order of magnitude.

Estimates of the annual input of these elements into San Francisco Bay from EBMUD effluent are presented in Table 57. These estimates are based on an average discharge rate of 3.04×10^8 liters/day or 1.1×10^{11} liters/yr and the mean concentrations of the elements observed in weekly composite sewage samples.

Table 57 Estimate of the annual input in metric tons/yr. of twelve elements into San Francisco Bay from sewage effluent discharged by the EBMUD Oakland sewage treatment plant. Values are based on 3.04×10^8 liters/day average flow rate and the mean concentrations of particulates and elements observed in eight, one-week composite sewage effluent samples collected between November, 1974 and March, 1975.

Element	Mean Concentration ug/l	Mean Annual Input to S. F. Bay Tonnes
Particulates	79200	8800
Ag	1.8	0.20
As	0.6	0.07
Cd	2.5	0.28
Cr	87	9.7
Cu	308	34
Fe	2300	255
Hg	0.7	0.07
Mn	39	4.4
Ni	14	1.6
Pb	111	12.3
Se	0.3	0.03
Zn	454	50

b. Chlorinated hydrocarbons in composite sewage samples

Three composite sewage samples were collected for chlorinated hydrocarbon analysis during December and again in February by the same

method described above. The samples were collected, however, in a hexane-rinsed all glass system and 200 ml of n-hexane were added to the samples at the time of collection to minimize possible biological activity during the periods of transfer and storage. Following the extraction of this mixture in a 2 liter separatory funnel, the sewage was re-extracted with two successive 200 ml volumes of 15% ethyl ether in hexane. Two hundred ml of a saturated aqueous solution of Na_2SO_4 were added to the samples prior to the final extraction. The ethyl ether in the combined extracts was removed by evaporation and the samples were subsequently treated as described in Part II, Section C-3 for water analyses. The results of these analyses are presented in Table A-38.

Past research has shown that sewage effluent can contribute a significant amount of chlorinated hydrocarbons to marine and estuarine ecosystems (Holden, 1970; Schmidt *et al.*, 1971; SCCWRP, 1973). High concentrations of PCB (as Aroclor 1254) were detected in the EBMUD sewage effluent and DDT compounds occurred in lesser amounts. Greater amounts of chlorinated hydrocarbons were observed in the sediments from the vicinity of the EBMUD outfall than in sediments from the Spoil Site area. Our results indicate that the PCB concentrations from the EBMUD plant were roughly equivalent to the levels found in effluent from the five mile outfall of the Hyperion treatment plant of the City of Los Angeles and in effluent from the San Diego Point Loma plant (SCCWRP, 1973).

E. Results of Water Chemistry Monitoring

1. Pre-spoil Period, 8-9 January, 1975

The results of the pre-spoil water chemistry monitoring are presented in Figure 6. Dissolved oxygen, nitrogen as ammonia, and pH appeared to correspond inversely with tidal height; higher D.O. and NH_3 concentrations and higher pH values were observed during low tide periods. Salinity values fluctuated in relation to the tidal period with slightly increased values occurring during high tide.

2. Spoiling Period, 3-5 February, 1975

The water chemistry data collected during the 42 hour disposal operation are presented in Figure 7. The temperature, salinity, pH, NH_3 , nitrate, and dissolved oxygen concentrations observed before the spoiling operation were similar to those observed during this collection period.

A decrease in dissolved oxygen and a slight increase in ammonia and nitrate levels were seen in samples collected from the spoil plume immediately following three of the disposals. These changes were of

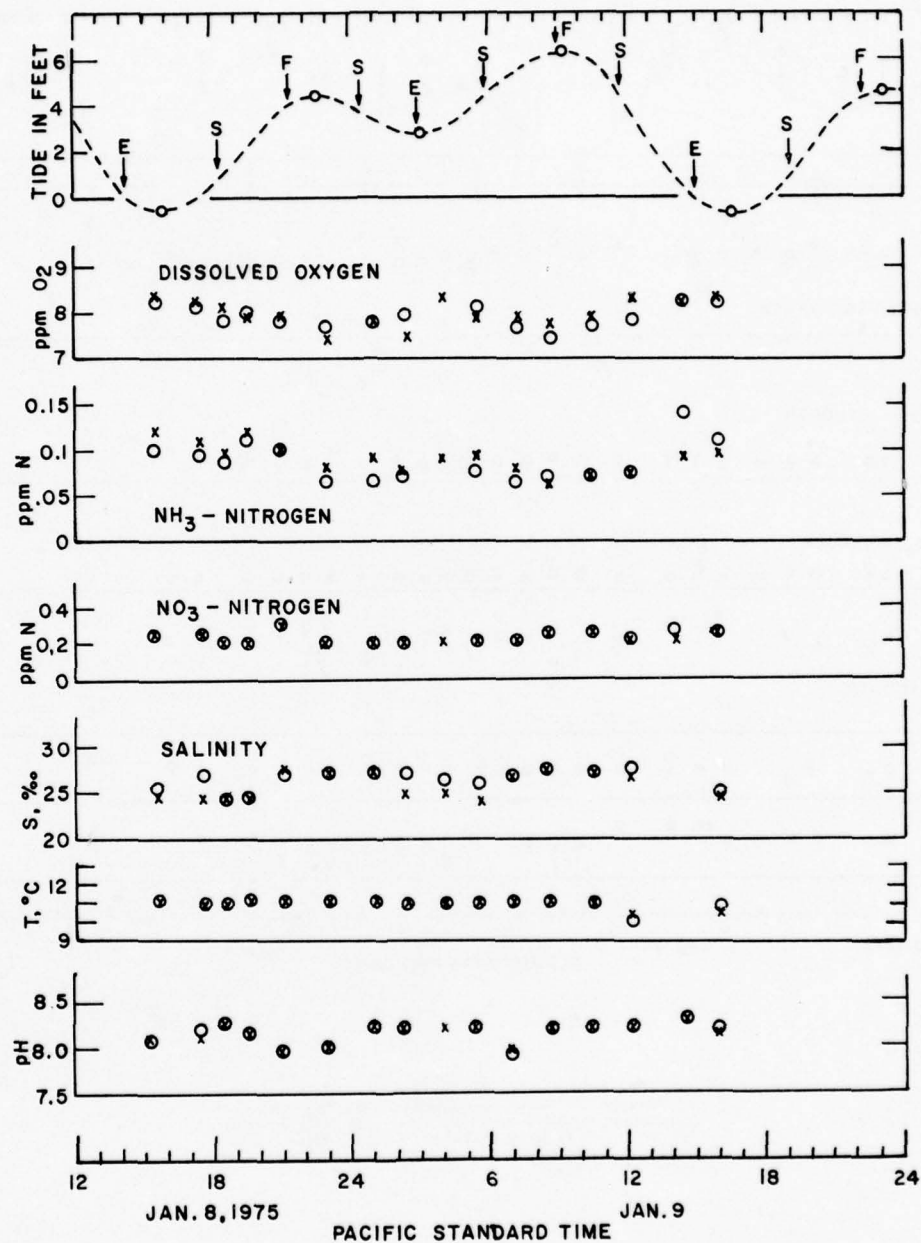


Figure 6. Results of water chemistry monitoring at Spoil Site stations 8-9 Jan. 1975, one month before the experimental spoiling operation. Measurements and collections taken hourly over a 24-hour period. x- top samples; O- bottom samples; E- maximum ebb tide; F- maximum flood tide; s- slack water.

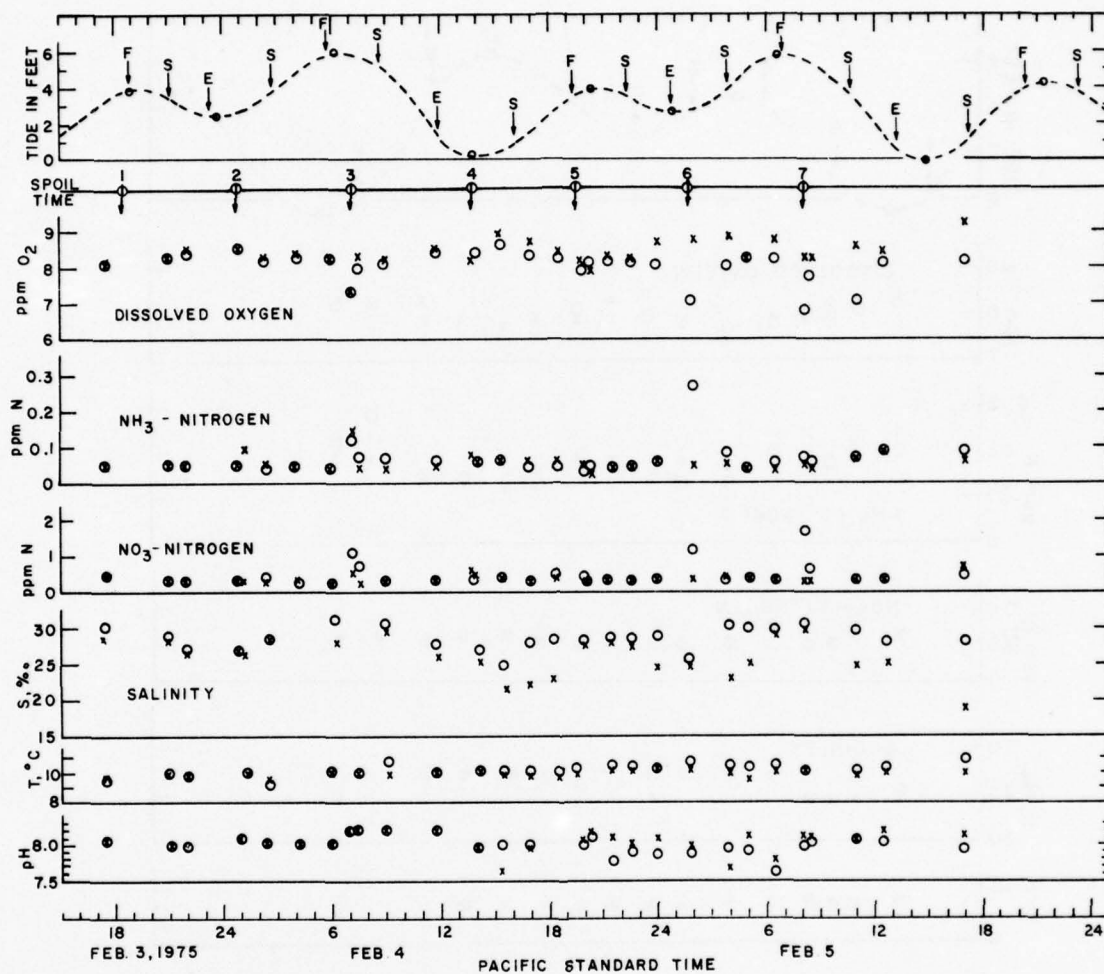


Figure 7. Results of water chemistry monitoring at Spoil Site stations 3-5 February, 1975 during the experimental spoiling operation. Measurements and collections taken hourly over the entire period and immediately following each of the seven disposals. x- top sample; o- bottom sample; E- maximum ebb tide; F- maximum flood tide; s- slack water.

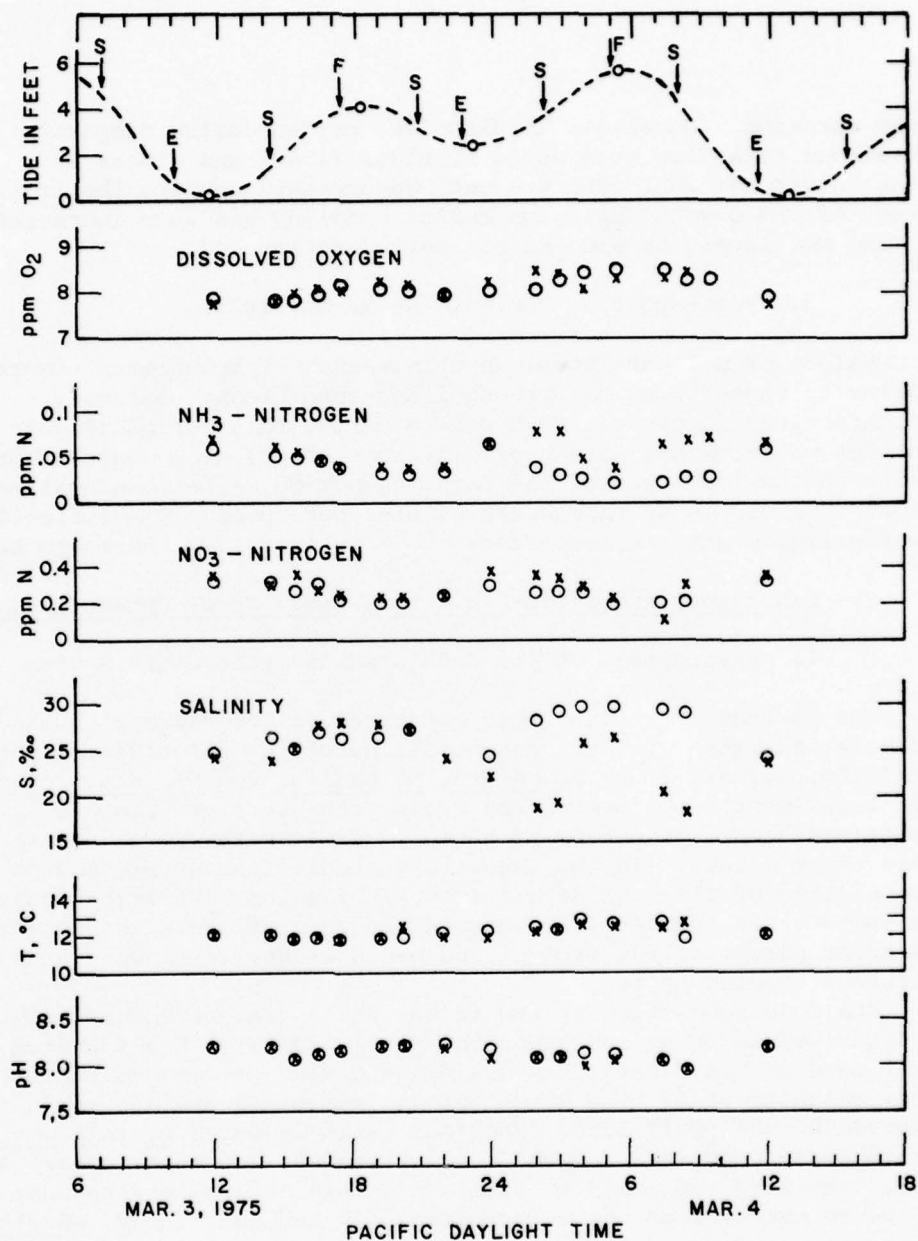


Figure 8. Results of water chemistry monitoring at Spoil Site stations 3-4 March, 1975; one month after the experimental spoiling operation. Measurements and water collections taken hourly over a 24 hour period. x- top sample; o- bottom sample; E- maximum ebb tide; F- maximum flood tide; s- slack water.

short duration. Decreases in dissolved oxygen during disposal operations have also been noted by Biggs (1968) and others as a transitory condition. The present measurements confirm the transitory nature of the oxygen depletion during spoiling and show increases in NH_3 and NO_3 levels to also be of short duration.

3. Post-spoiling Period, 3-4 March, 1975

Values of all parameters in this series of measurements were similar to those found for pre-spoiling conditions. Salinity concentrations fluctuated more widely during this period (Figure 8). Distinct water masses (areas of different color) were observed during the second twelve hours of the sampling period. The lower salinity values observed in surface water samples were probably a reflection of lower salinity water transported into this area from the north bay.

F. Laboratory and Field-Laboratory Heavy Metal Uptake Study

1. Description of the Bodega Marine Laboratory System

The laboratory studies were conducted to investigate uptake and accumulation rates of four concentrations of the chloride salts of the metals Cd, Hg, and Pb by M. nasuta, M. edulis, and P. californiensis. This experiment complimented the Field-Laboratory experiments (described below) in determining the possible pathways of uptake of these three metals. In the laboratory studies the uptake and accumulation of the test metals from water alone were investigated. The above three species were selected because of their use in nearly all other phases of the project and because they represent three different feeding types.

The continuous-flow system in which the organisms were maintained is described in detail by Anderlini et al. (1975). The filtered sea water used in the experiments was diluted with de-ionized water to a final salinity of 27 ‰ which was equivalent to average salinities observed at the Spoil Site. The four tanks in which P. californiensis were exposed contained a 4 cm layer of washed, #90 mesh, Ottawa silica sand. The sand was added to simulate a more natural environment and to insure survival of these burrowing polychaetes. The M. edulis and M. nasuta were suspended above the bottom of the tanks on Nytex screens mounted on Lucite strips. The screens isolated the animals from their fecal material and reduced coprophagy, especially by M. nasuta. Diluted sea water was added to each tank at a rate of 16 ml/min and the calculated turnover rate of each tank was approximately 10 hours. An air-injected venturi system was used to circulate and aerate the water.

Four exposure tanks were used per species. One tank served as a control and the others were used to expose the organisms to low, medium, and high concentrations of the three metal solutions. Three stock solutions were prepared with the following concentrations of the chloride salts of Cd, Hg, and Pb in 2 liters of 27 ‰ sea water:

	$\mu\text{g/l}$		
	Cd	Hg	Pb
Low	14.5	48.0	96.5
Medium	43.3	144.0	298.5
High	640.5	432.0	868.5

The stock solutions were siphoned through capillary tubing which metered the individual solutions into polyethylene bottles where they were vigorously mixed by Teflon stirring bars with filtered sea water in a 160:1 ratio. The "spiked" sea water was delivered to the appropriate tanks by Tygon overflow tubes. This system was assumed to provide the following final solution concentrations:

	$\mu\text{g/l}$		
	Cd	Hg	Pb
Control (assumed seawater concentrations based on Horne, 1969 and others)	$0.03 = X_1$	$0.1 = X_2$	$0.2 = X_3$
Low	$0.09 + X_1$	$0.3 + X_2$	$0.6 + X_3$
Medium	$0.27 + X_1$	$0.9 + X_2$	$1.8 + X_3$
High	$4.00 + X_1$	$2.7 + X_2$	$5.4 + X_3$

The spiked solutions were allowed to flow through the test tanks for three days before the animals were introduced. This time allowed the metals to adsorb onto and saturate the surfaces of the system and the sand in P. californiensis tanks. After three days it was assumed

that the system had reached a steady state with the metals in solution.

Approximately 680 P. californiensis and 460 M. nasuta of mixed sizes were collected from an area adjacent to Spoil Site station 1 and returned alive to the laboratory. Approximately 560 mussels of 25-35 mm in length were collected at the Berkeley Pier and cleaned of epifauna and also returned alive to the laboratory. All animals were placed in 27 ‰ filtered sea water to purge and acclimate for three days. At the end of this period (Day 0 of the experiment), three replicates of 15 P. californiensis, 13 M. nasuta, and 20 M. edulis each were removed and prepared for metals analysis as baseline samples. The animals were not fed during the three day purging-acclimation period or during the exposure experiments. The animals were observed daily to note behavior and mortality; the occasional dead animals were recycled.

At Day 0, 144 P. californiensis were placed in each of the four P. californiensis tanks, 120 M. edulis were placed in each M. edulis tank and 96 M. nasuta were placed in each M. nasuta tank. Each species was exposed to each of the four metal concentrations for 1, 3, 7, and 15 days. At the end of each of these periods, 36 P. californiensis, 30 M. edulis and 24 M. nasuta were removed from each tank and each group was divided into three equivalent replicates for metals analyses.

The flow rate through each tank was checked and calibrated three times per day. Salinity, pH, and temperature were measured every 24 hours. Salinity averaged $27.2 \pm .5$ ‰, while the average pH and temperature were $7.95 \pm .1$, $18.8 \pm 0.7^{\circ}\text{C}$, respectively. Twenty ml water samples for metals analyses were collected and composited daily from each mixing bottle overflow of each of the four concentrations. The four water samples, each containing 300 ml of water, were preserved by addition of 75 μ l of 10 M NBS 3 x distilled HCl to increase the pH to 2. Composite samples of each solution were collected to determine the average inflow concentration of the three metals during the period of study.

Water samples (250 ml) were collected from the outflows of tanks receiving low and high concentration water at the end of days 3 and 7 and preserved as above. Outflow water samples were collected to monitor actual exposure concentrations. The concentrations of the three metals in water were determined by the methods presented in Part II, Section B-5.

Sediment cores were collected from each of the four P. californiensis tanks at days 0, 7, and 15, and surface sediment samples were collected at day 15. All sediment samples were frozen in plastic vials for later analysis.

2. Results of Uptake Experiments

The results of the laboratory experiments are presented in Tables A-39 to A-41. These data were subjected to two-way analyses of variance the results of which are presented in Table 58.

Table 58 Laboratory Uptake Study. Resultant F ratios of a two-way analysis of variance model I of the difference in the observed concentrations of Cd, Hg and Pb in three species exposed for 1, 3, 7, and 15 days to water containing low, medium, or high concentrations of the test metals. Degrees of freedom equals (df).

Element			Cd	Hg	Pb
Species	Group	(df)			
<u>Macoma nasuta</u>	Time	(3)	1.16	201.0***	8.20**
	Concentration	(3)	13.63**	580.0***	39.92***
	Time X Conc.	(9)	.92	121.0***	18.22***
	Within subgroups (32)				
<u>Pectinaria californiensis</u>	Time	(3)	6.70**	7.76***	3.01*
	Concentration	(3)	1.47	53.38***	1.21
	Time X Conc.	(9)	1.77	2.99*	1.98
	within subgroups (32)				
<u>Mytilus edulis</u>	Time	(3)	44.76***	67.37***	42.17***
	Concentration	(3)	84.11***	130.76***	128.76***
	Time X Conc.	(9)	17.77***	17.37***	19.37***
	Within subgroups (32)				

* p<.05
 ** p<.01
 *** p<.001

Cadmium concentrations did not appear to increase significantly with time in M. nasuta although there was a significant increase in Cd concentrations in animals exposed to the highest concentrations. Concentrations of Cd in P. californiensis decreased significantly with time whereas there were no significant differences between P. californiensis exposure concentration groups. Highly significant increases in Cd concentrations were observed with time in M. edulis in low, medium and high concentrations. The highest concentrations of Cd were observed in mussels exposed to the highest concentrations, and the lowest concentrations were found in mussels in low exposure concentrations of Cd (Table A-39).

Cadmium concentrations in control M. nasuta and M. edulis were equivalent to the baseline concentrations observed in these species. However, Cd levels decreased with time in both control and test P. californiensis suggesting possible differences in the behavior of this element in these three species.

Highly significant differences in Hg levels between exposure concentration groups and through time were observed in all three species. Gradual increases in Hg levels through time were observed in the low exposure concentration groups and marked increases through time were seen in the highest exposure concentration groups. Mercury concentrations remained unchanged in M. nasuta and M. edulis control animals and appeared to decrease slightly in P. californiensis controls. Mercury levels increased ten-fold in M. nasuta and P. californiensis and almost twenty-fold in M. edulis exposed to the highest ambient Hg concentrations (Table A-40).

A significant difference in Pb concentrations with time was observed in all three species (Table 58). Significantly higher Pb concentrations were observed in M. edulis and M. nasuta exposed to the higher concentrations (Table A-41). A continuous increase in Pb levels through time was observed in M. edulis and M. nasuta exposed to the highest Pb concentrations. Lead levels in animals in the other ten tanks increased through day 3 and then decreased after 7 to 15 days exposure.

Except for Cd and Pb values in P. californiensis, and Cd values in M. nasuta, significant interactions between exposure times and concentrations were observed in all cases. These significant interactions were almost certainly due to the non-uniform timing of the exposure periods and the unequal increases in the concentrations used in this experiment.

Mytilus edulis and M. nasuta can accumulate Cd, Hg and Pb chlorides directly from water and P. californiensis can accumulate Hg and Pb but not Cd from water. Increases in Hg concentrations in all three species were proportional to exposure concentrations and linear increases in Hg concentrations were observed in P. californiensis. Linear increases in Cd concentrations were observed in M. edulis. Mercury and Cd concentrations remained constant and Pb levels decreased with time in M. edulis controls. All three elements remained constant with time in M. nasuta controls while Hg remained constant and Pb and Cd levels decreased with time in P. californiensis controls.

Composited inflow and outflow water samples were analyzed for Pb and Cd by the chelation-extraction method described in Part II, Section B-5a. Mercury in water was determined by the method described in Part II, Section B-5b. The results of these analyses and the results of Cd, Hg, and Pb determinations in sand samples collected from from P. californiensis tanks are presented in Tables A-42 and A-43 respectively.

Inflow concentrations show relatively close agreement with the assumed exposure concentrations of Cd, Hg, and Pb. Observed Pb values were 2-3 times higher than assumed and Cd levels were just slightly higher than expected (Table A-42). Mercury was the only metal measured in control sea water and the observed concentrations were 10 times lower than generally reported sea water values. This low Hg concentration agrees with the recent measurements of Hg in Puget Sound waters (Bothner and Robertson, 1975) and is similar to Hg concentrations observed in San Francisco Bay during the present study.

The results of the analyses of outflow water samples vary in their agreement with assumed concentrations of the three elements. Except for Cd values in the high concentration M. nasuta tanks, the observed outflow concentrations of Cd and Pb in M. edulis and M. nasuta tanks were not in agreement with inflow concentrations. The Cd and Pb concentrations observed in the outflow samples were up to an order of magnitude lower than inflow levels. These lower values could not be explained by the amount of uptake observed in the test animals. Therefore, sorption phenomena or some other physical-chemical processes may have resulted in the observed losses. It is also possible that these data are in error since the methods used in these analyses are subject to losses. The results of this experiment are believed to be valid since the uptake patterns observed in the test animals appeared to follow the order which could be predicted from the inflow water concentrations.

The sand collected from the four P. californiensis tanks after 0, 7, and 15 days did not appear to accumulate Hg, Cd, or Pb to concentrations within our detection limits. At the end of 15 days exposure, Cd concentrations were higher in surface sand samples than in sand cores collected at the same time. Lead concentrations in surface sand were higher than concentrations in sand cores in all but the medium concentration tanks. Mercury concentrations in surface sand were higher than concentrations in sand cores only in the high concentration tanks. The higher Cd and Pb concentrations observed in surface sand in P. californiensis tanks after 15 days could partially explain the lower values observed in outflow water from these tanks.

3. Description of Field-Laboratory System

Owing to the difficulties in designing and undertaking laboratory experiments which are representative of conditions in natural environments (Lee, 1973; Cronin et al., 1974), an in situ Field-Laboratory system was designed to isolate and examine the importance of the three pathways by which pollutants could be accumulated by selected organisms in San Francisco Bay.

The laboratory studies described above were performed to determine whether metals were accumulated directly from water. The Field-Laboratory was designed to expose invertebrates to altered amounts of suspended particulates and altered concentrations of pollutants in sediments. The pathways by which pollutants were accumulated in these organisms could then be investigated by exposing them to various combinations of altered conditions. However, the concentrations of pollutants associated with water, dissolved organic molecules, and very fine suspended particulates could not be controlled in this Field-Laboratory experiment.

The Field-Laboratory consisted of three 2.4 m long, by 1.2 m wide by .6 m high PVC plastic boxes (Figure 9). In use, the boxes were closed on the top and sides except for incurrent and excurrent ports at opposite ends. Each box had two doors on the top for installation and removal of transplant and benthic samples. The boxes were open on the bottom so they could be inserted into the bottom sediments forming a water-tight chamber above the sediment surface. Plastic skirts around the outside of the chambers prevented the side walls from sinking more than 15 cm into the sediment. Ribs with holes for attaching Nytex transplant bags were installed along the inside of the chamber top and transplant bags were attached with plastic shower curtain hooks.

The three chambers were connected to a 60 cm diameter Baker "Hydro" swimming pool filter fitted with an "Autotrol" self-flushing valve and powered by a one horsepower Peabody-Barnes submersible pump (Figure 10). Except for the cast iron casing and impeller on the sump pump, all fittings and plumbing were PVC plastic or Teflon. The "Autotrol" valve was encased in a specially constructed Lucite box. The filter unit was initially filled with #35 mesh Monterey sand.

Since this experiment involved testing benthic and intertidal invertebrates under natural conditions, it was necessary to locate the apparatus in an area with an abundant and diverse fauna and where the substratum was soft enough for the chambers to penetrate, but firm enough to support the 320 kg filtering unit. A site which was shallow enough for divers to work safely for extended periods and also out of heavily travelled shipping lanes was necessary. The area had to be large and homogeneous enough for the three chambers and the filtering unit to be moved about so that experiments could be repeated. A final requirement was the necessity of being near an electrical supply for the pump and automatic back flushing valve.

After several pre-survey collections, a suitable location was found on the south side of the Western Pacific Railroad's Mole Pier at the entrance to Oakland Inner Harbor (Figure 1). The study area was approximately 50 meters wide by 90 meters long with an average depth of 4 meters. In operation, water was drawn through the single up-channel

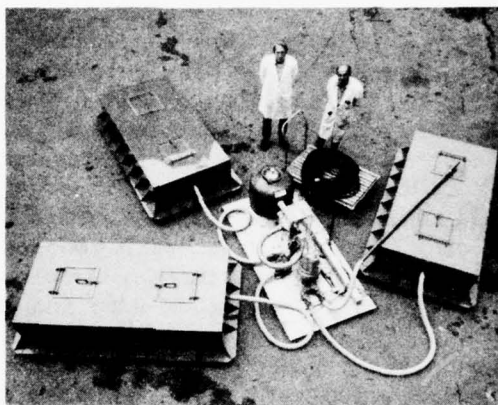


Figure 9. Field Laboratory mock-up showing the three laboratory boxes, the pump, the filter control unit, and the filter in the approximate arrangement they occupied in the field.

Lower left. - box with filtered water and local sediment. Upper left - box with filtered water and selected sediment. Center - pump and filter unit. Right - box with unfiltered water connected to intake to pump. A second intake is provided so flow can be equalized through all three boxes.

control chamber, through the pump and filtering unit, and then delivered in equal volumes to the two, downcurrent chambers. The flow rates were equalized through all chambers by PVC ball valves and an incurrent hose was installed alongside the upstream chamber to balance the flow through this tank. With #35 mesh sand, the filtering rate was estimated to be 60-120 liters/minute.

The chambers reduced the flow of water passing over the benthos and eliminated light reaching the bottom sediments. In addition suspended particulates were removed from the downstream chambers. Other than these alterations, the test conditions were similar to

natural conditions in this area. The conditions within the chambers were altered by reducing suspended particulate loads and could have been altered further by the addition of either clean or highly contaminated sediments. The concentrations of pollutants in organisms exposed to these altered conditions in the downstream chambers were to be compared with pollutant concentrations in organisms within the one control chamber and with organisms living outside of the chambers.

The Field-Laboratory experiments were conducted during October and November, 1974. In these experiments, samples of M. edulis and M. nasuta from the Oakland Inner Harbor site and M. edulis and Crassostrea gigas from Tomales Bay were exposed to ambient conditions, and to unfiltered and filtered sea water.

The mussels and oysters were each separated into 12 groups of 25 individuals each, and placed in Nylon transplant bags. Four bags each of local mussels and Tomales Bay mussels and oysters were placed within the unfiltered and filtered chambers and an additional four bags were hung outside of the chambers. Prior to transplanting, 25 individuals of each group were separated into 5 replicates of 5 animals each and prepared for trace element analyses as baseline samples.

Transplant animals were placed in the Field-Laboratory and removed after 1, 3, 7, and 15 days. At the end of each period, one bag of mussels and oysters from each group was removed and five replicates of 5 animals each from each group and collection were prepared for analysis.

4. Results of the Field-Laboratory Study

Trace element concentrations in mussels and oysters from the Field-Laboratory are presented in Table A-44. Concentrations of the twelve trace elements in each group were apparently not affected by the experimental conditions. No differences were found between groups or through time, and metal concentrations in mussels and oysters within the chamber receiving filtered water were identical to those observed in the other two groups.

Water samples were collected from within the unfiltered and filtered chambers and from an adjacent point outside of the chambers eight days after the transplant animals were placed in the Field-Laboratory. These samples were analyzed for total suspended particulate load and for trace element content of particulates (Table A-45).

A 35% reduction of suspended particulates was observed in water from the unfiltered chamber. The filter further reduced the suspended particulate load to less than 50% of ambient concentrations. These changes were greater than any of the fluctuations observed among Spoil Site collections (Tables A-31 and A-32).

No appreciable differences were observed in the concentrations of eight elements in these suspended particulate samples. The similar trace element composition of suspended particulates and the small reduction in particulate load produced by the filtering unit was a possible cause for the lack of any observable difference in the trace element concentrations of the test animals.

Various modifications to the Field-Laboratory filtering system were attempted to further increase the filtering efficiency. Several attempts to rectify the problems which developed in this equipment were unsuccessful and the experiment was terminated in late April, 1975.

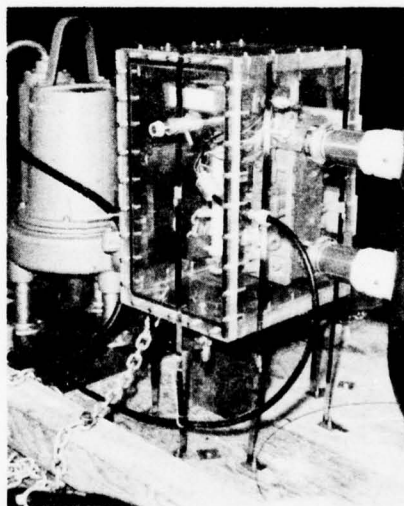


Figure 10 Field Laboratory filter control system. The control unit in a water-tight lucite box automatically switched valves to backflush the filter when the back pressure exceeded a preset value. The pump is at the rear left and the filter is to the right of the control unit.

PART IV. INTEGRATION AND DISCUSSION

A. The Effects of Spoiling Operations on Pollutant Availability

Trace Elements

Spoiling resulted in a short-term elevation of soluble Cd, Cu, and Pb concentrations at the impact center of the Spoil Site immediately following each of the seven disposals. Dissolved Pb, Cd, and Cu levels were 9, 6, and 4 times greater, respectively, in plume samples collected at the time of each disposal, than the average levels observed at other locations during this study. These higher concentrations at the center of the Spoil Site persisted less than 1.5 hours before being eliminated perhaps by mixing and dilution with tidal currents or by re-adsorption to suspended particulates. Similar increases in soluble Cd, Cu, and Pb concentrations were not observed in water samples collected at Spoil Site stations during the disposal operation.

In the laboratory study, three invertebrate species collected at the Spoil Site were exposed to concentrations of three metal salts similar to those observed at the Spoil Site prior to and during the disposal. Significant trace element accumulation directly from water required many hours to several days of continuous exposure. In addition, trace element losses from tissues of the animals used in the purge study and as laboratory controls occurred over a period of several days. Small, short-term increases in tissue concentrations were not detected in animals collected immediately after the disposal operation and no significant increases in trace element concentrations were observed in animals collected one month after spoiling. Therefore, spoiling operations, conducted under the conditions described, would not be expected to increase soluble trace element availability to invertebrates.

Copper and Fe concentrations in suspended particulates increased more at outer Spoil Site stations than at inner stations during spoiling. The increased Cu and Fe concentrations in suspended particulates during spoiling were not reflected in the concentrations of either of these two elements in invertebrates or sediments collected after spoiling. Therefore, the observed increase in Cu and Fe concentrations in suspended particulates during the disposal did not affect pollutant availability.

Except at station 2, trace element concentrations did not increase in settled particulates collected at the six Spoil Site stations. Spoiling accidentally occurred directly over station 2 which resulted in anomalously high deposition rates at this location. Deposition rates of settled particulates at the other Spoil Site stations after spoiling were not significantly greater than pre-spoiling rates. The

uniformity of deposition rates and of suspended particulate loads at Spoil Site stations following disposal operations indicate that unbiased, representative suspended particulate samples were obtained during spoiling.

The Field-Laboratory filtering system reduced suspended particulate loads by 50%. Trace element concentrations in mussels and oysters transplanted to the filtered water tanks were not significantly different from trace element concentrations in outside control animals and animals in unfiltered water. Undoubtedly, the filtering device preferentially removed the larger sized suspended particulates. Our results and those of Serner and Mercer (1975) show that an inverse correlation exists between particle size and trace element concentrations. Since no increases in particulate loads were detected at Spoil Site stations and since the greatest fraction of the spoil material was sand, no increases in element concentrations in adjacent filter feeding organisms would be expected during spoiling.

Although the concentrations of all elements were higher in dredge hopper sediments than in Spoil Site sediments, Cu and Fe were the only elements which increased more at inner stations after spoiling. Increased Cu and Fe concentrations in inner station sediments did not result in uptake of these elements in any invertebrate species analyzed. Therefore, pollutant availability from sediments was not affected by spoiling operations.

The slight increases in trace element concentrations in the invertebrate samples collected during this study were found to be significant. These increases, however, were unrelated to the spoiling operation; moreover, if even relatively small increases in trace element concentrations had occurred as a result of the spoiling operation, these would have been detected by the analytical system employed. If increases had occurred, these were of short duration and undetectable by the collection and analytical methods employed.

Chlorinated Hydrocarbons

Concentrations of chlorinated hydrocarbons in water from Spoil Site collections remained relatively stable throughout the study. The PCB concentrations were approximately an order of magnitude higher than those of either DDE or DDD. Significant increases in chlorinated hydrocarbon levels were observed in water samples collected immediately after several of the disposals, however: these increases persisted for less than one-half hour. Ten-fold increases in PCB concentrations and two to three-fold increases in p,p'-DDE levels were observed in some samples collected immediately after the disposals.

Chlorinated hydrocarbon concentrations fluctuated slightly in Spoil Site mussels during the period of study. Only p,p'-DDE levels

in M. edulis appeared to be affected by the spoiling operation. Concentrations of p,p'-DDE decreased in mussels at the outer Spoil Site stations and at both the Berkeley Pier and EBMUD stations after the disposal period, while p,p'-DDE concentrations in mussels at Spoil Site inner stations remained equal to pre-spoiling levels indicating that an increase occurred in the availability of this compound during the disposal operation. This effect did not persist in mussels after the spoiling period since, one month later, levels of DDE and other chlorinated hydrocarbons were equivalent at all stations (Table A-12).

The increased DDE concentrations in water, including suspended particulates, observed immediately following the disposal of dredged sediments could account for the persistence of DDE in inner station mussels. The short duration of these increases may also be a result of tidal flushing and/or adsorption onto settling particulates.

The water chemistry data, and suspended and settled particulate load data all indicate that relatively stable conditions existed at the Spoil Site during this study. No long term changes were observed.

B. Comparison of Trace Element and Chlorinated Hydrocarbon Inputs into San Francisco Bay

The estimated annual input of trace elements into S.F. Bay from the EBMUD sewage outfall is presented in Table 57. This estimate is based on average trace element concentrations observed in composite sewage samples during this study and the average daily outflow reported by EBMUD personnel.

Estimates of the gross redistribution of twelve trace elements in San Francisco Bay from dredging and spoiling operations are presented in Table 59. Redistribution due to the experimental spoiling operation was calculated from the total cubic yardage of sediment dumped at Spoil Site and the mean concentrations of trace elements observed in composite sediment samples from the dredge hoppers. Annual trace element redistribution from all dredging and spoiling operations in San Francisco Bay was estimated using a mean value of annual sediment dredged of $5.4 \times 10^6 \text{ m}^3$ of sediments (U.S. Army Corps of Engineers, 1975). Calculations were based on a measured moisture content of 50% and a specific gravity of 1.5. The mean concentrations of trace elements in Oakland Inner Harbor sediments were used as the estimate of trace element concentrations in all dredged sediments in the bay. This estimate is assumed to be a "worst case" estimate since Oakland Inner Harbor sediments are some of the most polluted in S.F. Bay (U. S. Army Corps of Engineers, 1975).

Estimates of the amount of deposition of trace elements from suspended particulates are presented in Table 60. Calculations for these estimates were based on the surface area of San Francisco Bay

Table 59. Estimates of the input of twelve trace elements into the Spoil Site area during the disposal of 10,000 m³ of Oakland Inner Harbor sediments and the estimated redistribution of trace elements from all dredging activity in San Francisco Bay. Estimates are based on a sediment density of 1.5^a), a 50% moisture content, the average disposal of 5.4 x 10⁶ m³/yr^b) of sediment, and average trace element concentrations observed in Oakland Inner Harbor sediments.

Element	Mean Concentration in Sediments ppm, dry weight (except Fe)	Spoil Site Input Tonnes	Annual Redistribution All S.F. Bay Spoiling Tonnes
Ag	1	.01	4.1
As	15	.09	60.8
Cd	1	.01	4.1
Cr	319	2.40	1,292.0
Cu	124	.93	502.2
Fe	5.0%	375.00	202,500.0
Hg	2.5	.02	10.0
Mn	710	5.33	2,875.5
Ni	260	1.95	1,053.0
Pb	117	.88	473.9
Se	3	.02	12.2
Zn	286	2.15	1,158.3

a) Measured wet density was 1.47.

b) Serne and Mercer, 1975.

Table 60 Estimate of the daily and annual depositions, in metric tons, of twelve trace elements associated with settling particulates in San Francisco Bay. Estimates are based on a surface area of 1166 sq. km, an average deposition rate of settled particulates of 1100 metric tons of sediment/km²/day and the grand mean of element concentrations (in ppm dry weight) observed in settled particulates from all stations and collections.

Element	Mean Concentrations in Settled Particulate ppm dry weight (except Fe)	Daily Input Tonnes	Annual Input Tonnes
Ag	0.9	1.2	4.38X10 ²
As	11.6	14.9	5.44X10 ³
Cd	0.9	1.2	4.38X10 ²
Cr	247.0	316.5	1.16X10 ⁵
Cu	72.0	92.3	3.37X10 ⁴
Fe	4.8	61,500.0	2.25X10 ⁷
Hg	0.5	0.7	2.56X10 ²
Mn	752.0	963.7	3.52X10 ⁵
Ni	228.0	292.2	1.07X10 ⁵
Pb	61.0	78.2	2.85X10 ⁴
Se	2.4	3.1	1.13X10 ³
Zn	171.0	219.1	8.00X10 ⁴

are valid within an order of magnitude since our estimates of suspended particulate loads agree with those of Pearson et al. (1970) and our estimates of trace elements in Bay sediments are also in agreement with other workers (McColloch et al., 1971; Peterson et al., 1972; Moyer and Budinger, 1974).

Although these input and redistribution estimates were based on trace element concentrations observed only during part of the year and on assumptions of uniform particulate depositions and average dredging activity, they provide an order of magnitude comparison of some of the sources of trace element input and transport in San Francisco Bay. By comparing the estimates presented in Tables 57, 59 and 60, it can be seen that settled particulates are the largest source of trace element redistribution. About 100 times the amount of trace elements redistributed by all dredging operations in San Francisco Bay is redistributed by settling particulates. The calculated amounts of trace elements redistributed by the two-day experimental disposal were roughly equivalent to the daily trace element redistribution by all dredging operations.

The annual amount of As, Fe, Mn, Ni, and Se introduced into San Francisco Bay by the EBMUD outfall is equivalent to the input of these elements during the two-day experimental disposal. The EBMUD outfall annually introduces much higher amounts of Cu, Pb, and Zn than is redistributed by a single day of dredging. However, the annual amount of trace element redistribution by dredging-spoiling activities is many times greater than the annual EBMUD input. Annual trace element redistribution by all dredging operations is almost inconsequential when compared with the annual amount of trace element movement associated with settling particulates.

Although these estimates are derived from many assumptions, they provide a gross means of comparing the magnitude of trace element redistribution within San Francisco Bay. In order to gain a better understanding of the magnitude of inputs of chlorinated hydrocarbons into San Francisco Bay, the various sources of contamination have been investigated, including the water, sewage effluent and spoil sediment measured in the present study. A recent study by the State Water Resources Control Board (1974) includes estimates of PCB's in non-urban runoff into San Francisco Bay of 20 ng/l, residential and commercial urban runoff of 300 ng/l and industrial runoff of 440 ng/l. Unfortunately, the source, location, volume of runoff or time of year are not indicated for any of these values, so they are of limited use in constructing mass emission estimated at 60 kg each of total DDT and PCB compounds in rainfall and 263 kg each of total DDT and PCB in dustfall. Thus, PCB emissions from aerial sources would total about 320 kg/yr. Again, the source of these values is not indicated, so their validity is not established.

Extrapolating from the limited data presented in this report a mean concentration of PCB in the water of San Francisco Bay of 1 ng/l would yield a total PCB load in the waters of the Bay of approximately 7.5 kg. This value assumes a mean tidal level and a total water volume of $6.65 \times 10^9 \text{ m}^3$. During the experimental dredge spoiling operations, in which 10,000 m^3 of sediment were dumped, it is estimated that approximately 0.8 kg of PCB entered the Bay, approximately 10% of the PCB present in the waters of the entire Bay (1165 km^2) at that time. The EBMUD outfall contributes in the order of 0.16 kg of PCB/day, approximately 20% of the PCB dumped during the spoiling operations. However, this is a daily input which would equal close to 60 kg/year, assuming that the PCB input is constant through the year. The EBMUD outfall is but one of many, so the total contribution of PCB from urban sewage outfalls probably is considerably higher than this estimate.

It is therefore apparent that a spoiling operation of the size examined in the present study would redistribute approximately 1% as much PCB as emitted annually from the EBMUD outfall. This amount (0.8 kg) compares to the estimated 320 kg deposited from aerial fallout annually and thus would be a relatively minor contribution to the total PCB burden in San Francisco Bay. If all of these estimates of PCB values are valid, PCB's must be very rapidly removed from the water column.

C. Recommendations

In the introduction, the authors presented factors which were considered to influence availability of pollutants to invertebrates in San Francisco Bay. These factors were: (1) the magnitude of pollutant inputs; (2) the rate of pollutant degradation or removal; (3) the physical and chemical characteristics of the receiving waters, including suspended particulates; (4) the physical and chemical characteristics of the respective pollutants; and (5) the physiological state of the organisms exposed to pollutants.

The magnitude of some pollutant inputs into San Francisco Bay has been estimated or measured by various state and federal agencies. Information is presently available for pollutant inputs from sewage outfalls, aerial fallout, dredging-spoiling operations and river transport. During the present study estimates of the annual input of trace elements and chlorinated hydrocarbons from the EBMUD sewage outfall were compared with estimates of pollutant redistribution by an experimental spoiling operation. Unfortunately, no accurate estimates of pollutant input from storm drains and surface runoff are available. This potentially large input source should be investigated so that more realistic estimates of total pollutant input can be made.

The dispersal rate of redistributed pollutants during spoiling operations appears to be very rapid. However, the net loss of

pollutants from San Francisco Bay has never been investigated. It is essential to gather information on the magnitude of pollutant input, the residence time of pollutants, and the net loss of pollutants so that accurate pollutant mass balance estimates can be constructed.

During the present study, physical and chemical characteristics of the receiving waters were only partially considered. Physical and chemical water parameters were measured at Spoil Site stations prior to, during, and after the disposal operation to monitor possible fluctuations in these parameters which may influence pollutant availability. Information concerning possible relationships among pollutants, organisms, and the physical-chemical state of ambient water is becoming increasingly available. Future field pollutant studies should include an investigation of ambient water characteristics so that physical or chemical factors which may control pollutant availability can be identified.

The physical and chemical state of the pollutants monitored was not determined in this study. The work of Luoma and Jenne (1975a, 1975b) and others have shown the importance of these factors on pollutant accumulation. The majority of these studies have been laboratory investigations. Future field studies of changes in pollutant concentrations in organisms in relation to changes in the physical or chemical characteristics of pollutants should be conducted.

Related to the physical-chemical state of pollutants are the pathways or routes by which pollutants may be accumulated by organisms. In the present study, we attempted to isolate three possible pathways (water, suspended particulates, and sediment) by which invertebrates may accumulate pollutants. In the laboratory study, uptake and accumulation of metal salts directly from water was demonstrated. Analyses of invertebrates collected at Spoil Site and EBMUD stations provide information on uptake and accumulation from sediment and suspended particulates. The "Field-Laboratory" was a direct attempt at isolating the three pathways in situ. The continuation of such in situ experiments appears to be warranted.

Collection methods employed during the present study took into account the physiological state of the invertebrates monitored. Compositing of samples and collection of organisms of a single size class was initiated to reduce variation among replicate samples. In the laboratory study, certain species accumulated pollutants passively while other species apparently regulated pollutant uptake. This may be true not only among species, but among individuals within different age groups of each species. It is well known that invertebrate larval forms are more sensitive than adults to environmental contamination. Our work and that of Boyden (1974) have shown that smaller individuals of some invertebrate species accumulate higher concentrations of trace elements than do larger individuals. Other studies and our own data have shown that different species accumulate different concentrations

of a particular pollutant and that the ratios of pollutants vary between species.

Few data exist concerning the dynamics of trace elements in relation to their availability to estuarine invertebrates, and the relationship of invertebrate physiology and biochemistry to pollutant uptake is not well understood. Most likely, the chemical state of elements, salinity, and other physical-chemical and biochemical factors synergistically influence the availability of each pollutant type. Identifying and investigating the major parameters which control pollutant availability in a system, such as San Francisco Bay, requires an understanding of the total pollutant flux through that system. Future studies should be designed to identify the parameters which govern pollutant flux through the entire system so that predictions of pollutant availability may be made over a wide range of conditions.

Data presented in this report suggest that long-term, seasonal fluctuations in pollutant concentrations in estuarine invertebrates may be controlled by only one or a few parameters. During this study, trace element concentrations generally increased with time in all invertebrates examined. Trace element concentrations also increased with time in surface sediments, while concentrations in settled particulates remained stable. The concentrations of soluble Cd, Cu, and Pb were equivalent in samples collected at Spoil Site stations during and one month after the disposal operations. Trace element concentrations in suspended particulates, however, decreased through time.

The mean concentrations of all elements in surface sediments, settled particulates, suspended particulates, benthic invertebrates and *M. edulis* from all collections were each subjected to a non-parametric multiple correlation analysis (Kendall's Coefficient of Concordance W) (Table 61). Highly significant correlations among all twelve elements in each of the above components were observed. This correlation suggests that trace element concentrations at the Spoil Site may have been controlled by one or a few parameters.

The data were also tested for significant relationships between element concentrations in these components and collection location. Only surface sediment concentrations varied significantly with stations. Trace element concentrations in benthic invertebrates were generally not correlated with those in surface sediments through time or between locations and were therefore not related to ambient sediment levels. Similar observations were made by Anderlini *et al.* (1975) and Halcrow and Mackay (1974). Furthermore, trace element concentrations in *M. edulis* were correlated with concentrations in benthic invertebrates suggesting that sediment element concentrations were not the primary factor controlling element levels in these species.

Concentrations of trace elements in settled particulates were not correlated with levels in any of the invertebrates analyzed. The

Table 61. Spoil Site. Kendall coefficient of Concordance W and χ^2 ratios for all trace elements and for all chlorinated hydrocarbons in each of the parameters examined. Degrees of freedom (df) are included.

	(df)	W	χ^2
Trace Elements			
Water		insufficient data	
Suspended Particulates	8	.45	42.86***
Settled Particulates	15	.26	46.88***
Surface Sediments	17	.33	68.25***
<u>Mytilus edulis</u>	23	.31	85.96***
<u>Macoma nasuta</u>	17	.37	74.98***
<u>Pectinaria californiensis</u>	17	.40	82.53***
<u>Stylatula elongata</u>	17	.35	70.97***
Chlorinated Hydrocarbons			
Water	31	.72	67.30***
Surface Sediments	11	.61	20.07*
<u>Mytilus edulis</u>	59	.59	104.87***
* $P < 0.05$ *** $P < 0.001$			

average size of settled particulates was greater than that observed in suspended particulates suggesting that the larger-sized settled particulates, at least, did not influence trace element concentrations in invertebrates. The Field-Laboratory findings also support this conclusion.

The concentrations of trace elements in suspended particulates decreased through time while concentrations in invertebrates increased. This suggests that element concentrations in suspended particulates did not influence element concentrations in these species. Concentrations of soluble Cd, Cu and Pb did not increase with time as observed in the invertebrates and therefore apparently did not directly affect concentrations of these metals in invertebrates. Soluble Pb concentrations observed in Mare Island Strait by Anderlini *et al.* (1975) also did not fluctuate in correlation with Pb concentrations observed in M. edulis from this area. These data and the Bodega Marine Laboratory studies indicate that very low concentrations of soluble Cd, Hg or Pb do not control the concentrations of these metals in invertebrate tissues.

Anderlini et al. (1975) and others have observed a negative correlation between trace element concentration in sediment and invertebrate tissues and the salinity of ambient water. This observation was supported by laboratory experiments with the clam, Macoma balthica, in which Anderlini et al. (op. cit.) observed much greater uptake of metal salts in animals exposed to low salinity than in animals exposed to high salinity water. Metal desorption in M. balthica was also much slower in low salinities. Trace element concentrations fluctuated more in animals less adapted to low salinities changes, such as Ampelisca milleri and M. edulis than in organisms such as Neanthes succinea and M. balthica which are known to be more euryhaline. In view of these observations, the gradual increases in trace element concentrations over time observed in sediments and invertebrates during the present study may be related to seasonal changes in the average salinity of central bay waters.

An alternative, and perhaps complimentary explanation for the gradual increases in trace element concentrations with time could be related to changes in the chemical states of the elements which were monitored. Luoma and Jenne (1975a; 1975b) found that the chemical state of trace elements can greatly affect the rate and amount of element uptake by the clam Macoma balthica. Since significant simultaneous changes in trace element concentrations were observed in all invertebrate species at the Spoil Site during a period of stable salinity, a change in the chemical state of the elements in water may be the best explanation for the observed results.

SUMMARY AND CONCLUSIONS

Summary

1. The concentrations of the trace elements Ag, As, Cd, Cr, Cu, Fe, Hg, Mn, Ni, Pb, Se, and Zn were monitored in sediments, suspended and settled particulates, selected benthic invertebrates, and mussels at an experimental disposal site in central San Francisco Bay, before, during, and after the disposal of 10,000 m³ of polluted sediments from Oakland Inner Harbor.
2. Dissolved Cd, Cu, and Pb concentrations were determined in centrifuged water samples collected during and after the experimental spoiling operation.
3. Concentrations of chlorinated hydrocarbons (DDT, DDD, DDE, and PCB, as Aroclor 1254) were also monitored in sediment, water including suspended particulates, and mussels at the experimental Spoil Site before, during and after disposal operations.
4. The water chemistry parameters, temperature, salinity, dissolved oxygen, nitrate, ammonia nitrogen, and pH were monitored

concurrently with water and suspended particulate collections during the 42 hour experimental disposal, and during 24 hour collection periods one month before and one month after the disposal operations.

5. Trace elements were monitored in sediments, settled particulates, benthic invertebrates and mussels at four stations adjacent to the East Bay Municipal Utility District sewage outfall during this study to compare the effects of a spoiling operation on the availability of pollutants with the effects of a large point source of pollutant input. Chlorinated hydrocarbon concentrations were monitored in sediments and mussels at these stations; trace elements and chlorinated hydrocarbons were determined in weekly composite sewage samples.
6. The pathways by which estuarine invertebrates may accumulate trace elements were investigated in a laboratory and in a "Field-Laboratory" study. The uptake of three concentrations of Cd, Hg, and Pb salts from water by three estuarine invertebrates (Macoma nasuta, Pectinaria californiensis, and Mytilus edulis) were investigated in a laboratory study at the Bodega Marine Laboratory. The influence of suspended particulates on trace element concentrations in estuarine invertebrates was investigated during a "Field-Laboratory" experiment in which selected invertebrates were exposed in situ to altered concentrations of suspended particulates.

Conclusions

1. Spoiling operations did not significantly affect trace element concentrations in benthic invertebrates at Spoil Site stations nor in M. edulis transplanted to this site. The spoiling operation did not therefore result in increased availability of trace elements.
2. Chlorinated hydrocarbon concentrations remained stable in transplanted mussels throughout the period of study. Only p,p'-DDE levels in M. edulis from Spoil Site inner stations appeared to be affected by the spoiling operation. Concentrations of p,p'-DDE decreased significantly more in mussels at Spoil Site outer stations, at Berkeley Pier and at EBMUD stations than in Spoil Site inner station mussels. This difference is possibly the result of increased DDE availability during the disposal period. However, this effect did not persist after the spoiling period; one month later, levels of DDE and other chlorinated hydrocarbons were equivalent at all stations.
3. Copper and Fe concentrations in surface sediments increased more at Spoil Site inner stations than at outer stations after the disposal operations. Since a similar increase was not noted at

- EBMUD stations, the spoiling operation apparently resulted in increased levels of Cu and Fe at Spoil Site inner stations.
4. Copper and Fe concentrations in suspended particulates appeared to be affected by the spoiling operation. However, the significance of the observed results is difficult to interpret since Cu and Fe concentrations increased more in suspended particulates collected at outer Spoil Site stations than at inner stations. No adequate explanation for the observed results is available.
 5. Spoiling resulted in a short-term elevation of dissolved Cd, Cu, and Pb concentrations at the impact center of the Spoil Site immediately following each of the seven disposals. These increases persisted for less than 1.5 hours and were probably eliminated either by mixing and dilution by tidal currents or by adsorption to suspended particulates.
 6. Short-term (less than 0.5 hour) increases were also observed in chlorinated hydrocarbon concentrations in water samples collected within the spoil plume immediately following several of these disposals. Ten-fold increases in PCB concentrations and two to three-fold increases in DDE were observed in some water samples collected immediately after the disposals as compared with other samples collected during this study.
 7. Water chemistry data and suspended and settled particulate load data all indicate that the water quality conditions remained stable during the period of study.
 8. At the Spoil Site, trace element concentrations generally increased with time in all invertebrate species examined. Element concentrations increased with time in surface sediments, while concentrations in settled particulates remained stable. Concentrations of soluble Cd, Cu, and Pb at Spoil Site stations during the disposal period were equivalent to levels measured one month later. Trace element concentrations in suspended particulates decreased with time.
 9. Concentrations of twelve trace elements fluctuated in coincidence with each other in sediments, invertebrates, and settled and suspended particulates. This suggests that one or only a few parameters may have controlled trace element concentrations at the Spoil Site during the study period.
 10. Trace element concentrations in sediments were not correlated with trace element concentrations in benthic invertebrates. Trace element concentrations in ~~settled~~ particulates were not correlated with concentrations in any of the invertebrates analyzed. Since concentrations in suspended particulates decreased through time, they were not correlated with levels in invertebrates or in sediments. Dissolved Cd, Cu, and Pb concentrations were also not correlated with concentrations of these elements in any of the above components of the estuarine system.

11. Concentrations of all twelve elements in M. nasuta at EBMUD stations were higher than in M. nasuta from Spoil Site stations. Trace element concentrations in P. californiensis, G. americana, and G. robusta were equivalent at both areas.
12. DDE and PCB concentrations were 2 and 3 times higher, respectively, in EBMUD sediments than in Spoil Site sediments, whereas DDD levels were equivalent in the two sediments.
13. Trace element concentrations were similar in mussels at Spoil Site, EBMUD, and Berkeley Pier. A gradual increase in element concentrations was observed at all three sites indicating a general increase in trace element availability in central S.F. Bay between Dec. 1974 and Mar. 1975.
14. DDD and PCB levels increased with time in mussels from all three sites. The greatest increases were observed at EBMUD stations. DDE concentrations decreased with time in mussels at all three sites with the greatest decreases noted at EBMUD stations. Chlorinated hydrocarbon levels were generally equivalent in mussels from all three sites during the period of study.
15. In the laboratory studies of uptake of metals from sea water:
 - (a) Mercury in solution was accumulated from sea water at all concentrations by M. nasuta, P. californiensis and M. edulis.
 - (b) Little, if any, Pb was accumulated by P. californiensis from sea water. Control animals of this species desorbed lead during the 15 day experiment. A slight uptake of Pb was observed in M. nasuta when these animals were exposed to the highest Pb concentrations tested. Pb was accumulated by M. edulis at all concentrations tested with desorption occurring in the control mussels.
 - (c) Cadmium concentrations decreased in P. californiensis at all concentrations tested. Cadmium was accumulated by M. nasuta exposed to high and medium concentrations, with the Cd content of control animals remaining constant throughout the experiment. M. edulis accumulated Cd linearly with time at all exposure concentrations. The Cd content of control mussels was constant throughout the experiment.
16. Estimates of the gross input of trace elements into San Francisco Bay during the experimental disposal operations were compared to inputs from all dredging-spoiling activity in S. F. Bay, inputs from the EBMUD sewage outfall, and estimated amounts of trace elements associated with settling particulates. In this comparison, by far the largest amounts of trace elements are associated with settling particulates. The daily transport of trace elements associated with settling particulates is approximately 100 times greater than transport resulting from a single day of all dredging activity in S.F. Bay. The amount of trace elements redistributed by the experimental disposal operation was equivalent to the average daily redistribution of

trace elements by all S.F. Bay dredging. The amount of trace elements redistributed annually by all dredging activities is much greater than the annual input from the EBMUD outfall but almost inconsequential in relation to element redistribution by settling particulates.

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APPENDIX I-A

TABLES OF DATA

Table A-1. Spoil Site. Trace element concentrations in parts per million dry weight in Macoma nasuta during collections I (19 Dec. 1974), II (15 Jan. 1975) and III (20 Feb. 1975). Three replicates, one standard deviation (), 10-20 individuals per replicate.

Elements	Collection	Stations					
		1	2	3	4	5	6
Ag	I	13.5 (1.5)	12.2 (2.0)	10.1 (0.7)	12.0 (2.0)	9.6 (3.1)	11.5 (2.4)
	II	14.7 (2.6)	13.0 (1.6)	12.1 (4.0)	8.9 (1.6)	13.6 (3.7)	18.3 (2.6)
	III	12.5 (2.7)	11.1 (1.8)	13.7 (2.0)	16.2 (3.2)	14.6 (2.3)	14.5 (10.3)
As	I	14.4 (1.4)	16.8 (2.2)	15.5 (3.6)	16.9 (0.7)	14.7 (1.3)	13.7 (0.2)
	II	17.3 (1.3)	19.2 (1.5)	17.7 (2.0)	15.0 (1.4)	18.1 (2.9)	17.7 (2.8)
	III	15.3 (0.9)	15.2 (1.7)	16.3 (2.1)	17.1 (2.3)	15.1 (0.2)	18.2 (1.8)
Cd	I	2.1 (0.3)	1.9 (0.3)	1.3 (0.1)	1.2 (0.6)	1.0 (0.4)	1.4 (0.7)
	II	1.8 (0.1)	0.8 (0.3)	0.7 (0.3)	0.6 (0.2)	1.5 (0.1)	1.7 (0.7)
	III	1.1 (0.5)	1.4 (0.6)	1.5 (0.5)	1.8 (0.3)	1.4 (0.8)	1.3 (0.3)
Cr	I	4.7 (1.2)	8.3 (8.7)	5.1 (4.0)	4.2 (2.9)	4.4 (2.1)	5.8 (1.0)
	II	9.8 (6.3)	8.6 (2.5)	5.7 (1.3)	8.7 (0.2)	3.8 (2.0)	9.2 (2.2)
	III	5.3 (1.0)	6.1 (1.2)	8.4 (2.8)	5.1 (1.9)	8.9 (4.0)	7.0 (3.5)
Cu	I	33.9 (4.2)	37.9 (2.4)	35.5 (2.3)	37.2 (4.7)	31.8 (6.8)	29.3 (4.0)
	II	42.8 (11.8)	40.9 (4.5)	40.2 (7.6)	36.2 (4.1)	44.9 (9.8)	43.3 (8.7)
	III	39.3 (5.4)	37.8 (3.1)	40.3 (2.7)	42.8 (7.6)	41.6 (6.1)	40.1 (7.0)
Fe	I	492 (39)	775 (231)	656 (45)	559 (30)	498 (32)	467 (3)
	II	861 (446)	924 (131)	652 (92)	683 (47)	736 (140)	805 (47)
	III	609 (173)	593 (72)	645 (77)	777 (147)	877 (178)	876 (134)

Table A-1 continued.

Element	Collection	Stations					
		I	2	3	4	5	6
Hg	I	.39 (.01)	.37 (.05)	.26 (.01)	.34 (.06)	.34 (.07)	.40 (.04)
	II	.46 (.06)	.42 (.11)	.40 (.04)	.38 (.02)	.40 (.04)	.46 (.03)
	III	.42 (.05)	.42 (.03)	.33 (.04)	.47 (.05)	.44 (.02)	.36 (.04)
Mn	I	20.8 (2.3)	23.1 (3.7)	25.7 (1.7)	23.0 (1.2)	16.5 (2.3)	15.8 (1.4)
	II	28.3 (9.4)	26.1 (1.1)	20.5 (3.2)	24.7 (.9)	20.9 (4.6)	25.3 (4.0)
	III	24.6 (2.2)	22.9 (4.8)	26.7 (3.0)	24.4 (2.4)	28.5 (4.4)	24.9 (4.7)
Ni	I	7.8 (1.5)	8.0 (2.1)	7.2 (1.2)	9.3 (1.4)	7.0 (1.9)	7.1 (1.6)
	II	11.0 (1.9)	8.5 (1.0)	8.4 (0.7)	7.5 (0.9)	10.1 (2.7)	10.2 (2.1)
	III	6.6 (0.3)	7.3 (1.1)	6.5 (1.1)	6.4 (0.4)	7.3 (1.0)	8.6 (2.2)
Pb	I	6.0 (0.4)	5.7 (2.0)	4.9 (0.3)	4.8 (0.8)	4.3 (0.1)	4.4 (1.3)
	II	4.9 (1.2)	5.8 (0.6)	5.5 (1.0)	5.6 (0.1)	5.5 (1.1)	5.2 (1.0)
	III	4.9 (0.7)	6.0 (2.4)	4.6 (0.5)	4.9 (0.8)	5.5 (1.9)	5.1 (0.7)
Se	I	3.7 (0.2)	3.7 (0.3)	4.4 (0.6)	4.2 (0.1)	3.4 (0.3)	3.7 (0.8)
	II	4.5 (0.2)	4.6 (0.8)	4.1 (0.3)	3.7 (0.5)	3.8 (0.2)	4.1 (0.2)
	III	4.0 (0.5)	4.0 (0.7)	4.6 (0.5)	4.4 (0.6)	3.6 (0.1)	4.6 (0.3)
Zn	I	284 (102)	261 (53)	250 (10)	309 (57)	248 (50)	224 (44)
	II	309 (64)	314 (20)	282 (46)	275 (54)	255 (63)	258 (36)
	III	247 (4)	283 (81)	307 (29)	301 (23)	222 (25)	285 (96)

Table A-2. Spoil Site. Trace element concentrations in parts per million dry weight (except Fe) in *Pectinaria californiensis* during collections I (19 Dec. 1974), II (15 Jan. 1975) and III (20 Feb. 1975). Three replicates, one standard deviation (), 10-20 individuals per replicate.

Element	Collection	Stations					
		1	2	3	4	5	6
Ag	I	3.5 (.5)	2.7 (.4)	2.6 (.6)	3.2 (.5)	2.6 (.7)	3.2 (.7)
	II	3.6 (1.4)	4.5 (1.5)	2.9 (1.0)	3.1 (.6)	4.0 (.4)	4.6 (.5)
	III	3.5 (.9)	3.6 (1.0)	3.6 (.01)	4.2 (.5)	4.0 (.5)	3.8 (1.2)
As	I	9.8 (1.4)	11.6 (.8)	10.9 (1.1)	12.3 (1.0)	11.2 (.3)	11.5 (3.0)
	II	9.9 (1.0)	11.1 (1.0)	9.4 (1.5)	12.3 (.9)	9.9 (1.2)	11.4 (1.6)
	III	12.8 (1.2)	10.5 (1.4)	14.8 (2.2)	12.2 (3.0)	11.1 (.5)	12.9 (.6)
Cd	I	2.8 (.8)	1.6 (.3)	2.5 (.7)	3.7 (.2)	2.8 (.4)	3.3 (1.0)
	II	3.2 (.9)	3.7 (.4)	2.5 (.6)	3.2 (.5)	3.9 (.4)	4.4 (.6)
	III	3.1 (.2)	1.7 (1.0)	3.1 (.9)	4.6 (.9)	3.1 (.5)	3.2 (1.1)
Cr	I	27.7 (5.2)	7.1 (2.8)	22.5 (2.0)	19.3 (4.0)	15.7 (2.9)	23.4 (17.7)
	II	18.6 (12.0)	29.3 (7.6)	39.2 (17.4)	10.0 (3.7)	8.0 (2.6)	26.7 (7.1)
	III	45.4 (10.5)	40.3 (5.4)	12.0 (2.5)	56.5 (7.0)	92.7 (14.0)	41.8 (21.0)
Cu	I	26.9 (1.8)	26.6 (2.0)	20.2 (3.4)	22.9 (2.0)	29.5 (3.0)	23.2 (3.4)
	II	34.2 (3.1)	33.8 (4.5)	34.5 (3.8)	31.1 (6.1)	22.0 (.7)	28.9 (1.6)
	III	41.7 (5.2)	34.2 (2.6)	33.7 (2.6)	33.8 (3.8)	31.0 (2.8)	35.2 (1.9)
Fe %	I	.40 (.08)	.21 (.03)	.40 (.07)	.38 (.05)	.24 (.04)	.37 (.10)
	II	.21 (.13)	.38 (.05)	.43 (.17)	.21 (.07)	.16 (.01)	.40 (.09)
	III	.49 (.15)	.52 (.14)	.31 (.03)	.63 (.17)	1.0 (.14)	.48 (.16)

Table A-2 Continued

Element	Collection	Stations					
		1	2	3	4	5	6
Hg	I	.14 (.01)	.15 (.17)	.14 (.02)	.14 (.02)	.14 (.03)	.13 (.05)
	II	.15 (.01)	.15 (.02)	.17 (.02)	.12 (.01)	.15 (.03)	.16 (.02)
	III	.20 (.01)	.18 (.01)	.14 (.01)	.19 (.01)	.17 (.01)	.17 (.00)
Mn	I	48.7 (14.8)	18.2 (4.4)	36.4 (7.9)	28.0 (11.5)	25.1 (5.4)	29.3 (13.7)
	II	42.7 (26.6)	43.3 (4.9)	83.0 (32.8)	15.9 (10.0)	9.0 (2.7)	38.2 (9.5)
	III	62.9 (3.7)	64.4 (25.5)	23.5 (3.4)	82.0 (25.5)	123 (24.6)	61.3 (14.8)
Ni	I	10.8 (2.7)	4.1 (.5)	9.2 (2.0)	5.9 (.2)	7.0 (1.7)	6.5 (3.5)
	II	9.5 (4.3)	10.8 (2.2)	17.4 (7.4)	4.8 (3.5)	3.9 (.5)	10.0 (2.4)
	III	13.1 (3.4)	19.0 (2.6)	7.3 (2.3)	17.4 (3.6)	28.1 (7.3)	15.0 (5.1)
Pb	I	6.6 (1.3)	7.4 (1.4)	6.0 (1.3)	6.4 (.6)	5.0 (.3)	5.8 (.4)
	II	10.4 (.7)	7.1 (1.3)	13.0 (.2)	5.4 (.8)	4.5 (1.2)	4.8 (.6)
	III	6.9 (2.7)	10.0 (1.2)	11.3 (.9)	9.0 (1.0)	7.7 (.3)	9.5 (.4)
Se	I	8.7 (.4)	8.9 (.01)	8.1 (.9)	8.7 (.2)	9.2 (.8)	10.5 (2.3)
	II	12.6 (.5)	9.0 (.5)	12.7 (1.3)	9.6 (.7)	7.1 (.4)	8.4 (1.1)
	III	9.4 (.3)	10.0 (1.3)	11.3 (.9)	9.0 (1.0)	7.7 (.3)	9.5 (.4)
Zn	I	91.1 (7.7)	91.8 (6.4)	82.8 (9.7)	89.4 (5.0)	101 (1.1)	92.1 (15.5)
	II	120 (10.7)	115 (10.2)	108 (11.3)	99.2 (10.4)	84.9 (3.9)	95.7 (10.4)
	III	130 (8.4)	128 (9.0)	133 (9.4)	125 (8.6)	129 (8.5)	138 (4.7)

Table A-3. Spoil Site. Trace element concentrations in parts per million dry weight in *Stylatula elongata* during collections I (19 Dec. 1974), II (15 Jan. 1975) and III (20 Feb. 1975). Three replicates unless otherwise indicated, one standard deviation (), 20 individuals per replicate.

Element	Collection	Stations					
		1	2	3	4	5	6
Ag	I	<1 a)	2.1 a)	1.7 (1.6)	2.6 (.2)	2.3 (.2)	1.6 (1.7)
	II	1.4 (.05)	1.3 (.2)	2.1 (.8)	1.9 (1.2)	1.2 (.7)	2.1 (.9)
	III	2.3 (.9)	1.8 (1.2)	1.8 (.3)	1.2 (.2)	2.6 (.9)	1.8 (.7)
As	I	26.7	25.9	31.8 (.3)	41.9 (3.6)	52.9 (5.6)	37.5 (7.4)
	II	37.1 (3.3)	46.5 (11.3)	44.6 (15.4)	46.3 (6.6)	42.0 (1.0)	56.1 (5.9)
	III	43.5 (1.3)	51.0 (5.0)	49.1 (3.0)	51.4 (3.7)	45.1 (2.5)	49.9 (.3)
Cd	I	1.3	5.1	3.4 (1.2)	3.7 (.6)	2.8 (.4)	3.9 (2.0)
	II	3.7 (.7)	3.1 (.8)	3.3 (1.6)	2.9 (.3)	2.4 (1.2)	3.7 (1.7)
	III	4.0 (.2)	3.6 (.6)	3.5 (.6)	3.4 (.8)	3.7 (1.1)	3.4 (.5)
Cr	I	1.3	6.5	8.4 (4.9)	10.5 (6.6)	8.0 (4.8)	10.9 (6.7)
	II	14.9 (1.1)	7.1 (2.0)	4.2 (.8)	10.4 (3.7)	10.6 (5.0)	12.3 (3.4)
	III	15.6 (1.2)	12.7 (.6)	10.0 (4.4)	17.2 (1.9)	7.1 (1.4)	8.1 (.3)
Cu	I	5.4	9.9	8.6 (1.1)	7.8 (.2)	8.3 (1.8)	8.0 (1.6)
	II	9.4 (1.5)	10.3 (2.4)	8.0 (.8)	9.0 (.4)	7.6 (.1)	11.1 (4.3)
	III	10.6 (.1)	8.2 (1.0)	8.4 (.4)	10.7 (1.8)	8.9 (1.9)	9.3 (1.0)
Fe	I	740	812	1175 (45)	933 (116)	1173 (62)	1126 (64)
	II	1057 (46)	869 (21)	1073 (68)	1186 (79)	1324 (55)	1512 (180)
	III	1646 (42)	1933 (138)	2287 (230)	2037 (164)	1625 (195)	1602 (99)

a) replicate number is one for samples at stations 1 and 2 collection I

Table A-3 continued

Element	Collection	Stations					
		1	2	3	4	5	6
Hg	I	.08 ()	.08 ()	.12 (.06)	.06 (.01)	.11 (.04)	.10 (.02)
	II	.08 (.01)	.08 (.02)	.07 (.01)	.07 (.01)	.08 (.01)	.07 (.01)
	III	.13 (.06)	.10 (.03)	.10 (.01)	.13 (.02)	.08 (.01)	.11 (.01)
Mn	I	6.0 ()	8.7 ()	14.2 (2.8)	16.0 (4.6)	16.8 (.7)	20.9 (5.7)
	II	7.5 (2.7)	13.7 (4.1)	18.1 (7.0)	21.1 (5.0)	22.0 (1.7)	21.3 (2.4)
	III	21.4 (4.9)	23.3 (3.7)	28.4 (3.7)	24.2 (2.9)	21.2 (1.4)	21.8 (3.8)
Ni	I	2.6 ()	4.6 ()	4.6 (1.2)	6.2 (2.6)	4.0 (.5)	4.5 (3.3)
	II	5.4 (1.8)	2.9 (1.0)	3.2 (.4)	3.4 (.4)	2.9 (.8)	5.7 (1.9)
	III	3.8 (.6)	5.9 (1.9)	4.9 (.8)	5.5 (2.6)	3.4 (1.4)	3.1 (.3)
Pb	I	1.8 ()	6.4 ()	3.2 (.6)	3.4 (1.0)	3.9 (.7)	4.4 (1.1)
	II	4.3 (1.6)	4.9 (1.4)	3.2 (1.0)	5.7 (1.1)	3.6 (.8)	4.1 (1.3)
	III	3.1 (1.0)	3.3 (1.3)	3.3 (.9)	3.9 (1.3)	3.3 (2.5)	3.3 (1.1)
Se	I	3.8 ()	3.3 ()	4.4 (1.0)	4.9 (.3)	5.0 (.4)	4.8 (1.0)
	II	4.5 (.3)	4.6 (.8)	4.6 (.4)	5.2 (.4)	4.4 (.6)	5.2 (.8)
	III	4.2 (.3)	5.2 (.2)	5.3 (.5)	4.9 (.5)	5.1 (.6)	5.8 (.7)
Zn	I	356 ()	314 ()	367 (15)	361 (41)	366 (16)	345 (6)
	II	350 (32)	332 (7)	320 (45)	345 (53)	371 (8)	368 (23)
	III	448 (33)	503 (40)	518 (32)	523 (50)	461 (58)	478 (34)

Table A-4. Table A-4. Spoil Site. Trace element concentrations in parts per million dry weight in miscellaneous invertebrates from collections I (19-20 Dec. 1974), II (25-26 Jan. 1975) and III (19-20 Feb. 1975). Means of \bar{n} replicates, 1 standard deviation (), 5-20 individuals per replicate.

Element:		Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
Species	Collection n												
MOLLUSCA:													
<i>Tritonia</i>	I 5	1.6(.7)	6.9(2.3)	29.1(7.7)	5.1(1.9)	5.7(2.1)	1187(292)	.13(.04)	12.8(9.2)	1.4(.9)	5.8(2.3)	7.9(1.2)	266(56)
<i>dimedia</i>	II 5	1.6(.6)	6.8(1.2)	25.2(4.1)	3.1(3.2)	6.4(2.5)	1057(451)	.12(.03)	10.7(3.4)	1.2(.5)	9.5(11.9)	6.8(1.3)	258(57)
	III 3	2.0(.9)	7.3(2.5)	29.7(3.2)	5.9(2.9)	6.2(1.8)	965(189)	.19(.04)	17.4(4.0)	3.1(.7)	6.8(2.5)	9.5(2.5)	298(67)
<i>Arminia</i>	II 2	1.4(.2)	5.8(.2)	19.1(13)	7.7(6.8)	6.0(.5)	1990(878)	.09(.01)	46.0(2.1)	3.5(2.6)	8.1(4.2)	8.1(4.5)	389(57)
<i>californica</i>													
<i>Solen</i>	II 2	3.6(.5)	4.5(.3)	3.0(.6)	1.9(2.4)	25.3(4.0)	200(15)	.15(.02)	7.8(2.2)	2.1(1.1)	2.3(.2)	4.1(.4)	80(.5)
<i>nicarius</i>	III 1	2.8	5.2	1.8	7.2	21.6	537	.23	10.5	3.6	2.0	4.6	102
<i>Siliqua</i>	II 1	3.8	6.8	2.5	1.7	9.8	990	.16	16.2	2.5	5.2	6.6	79
<i>patula</i>	III 1	6.3	7.0	1.9	17.0	16.9	2086	.25	35.0	5.8	3.1	6.4	89
<i>Cryptomya</i>	III 1	1.6	9.3	2.0	16.7	20.2	1714	.15	32.6	7.9	3.9	8.4	99
<i>californica</i>													
<i>clinocardium</i>	III 1	<1	8.0	<1	4.3	13.7	808	.28	27.8	12.9	3.8	6.1	182
<i>nuttalli</i>	III 1	4.7	5.0	4.3	7.0	24.6	306	.20	8.5	6.1	4.1	5.0	86
<i>Saxidomus</i>	III 1	3.4	8.1	2.9	6.9	23.0	455	.27	31.9	5.8	5.5	6.0	128
<i>nuttalli</i>													
<i>Comptosia</i> sp.	III 2	9.6(.8)	24.1(1.1)	6.4(1.6)	10.2(5.9)	13.0(.5)	1395(932)	.90(.18)	20.2(7.2)	11.2(3.3)	15.9(6.2)	9.6(.1)	172(76)
<i>Tapes japonica</i>	III												

Table A-4 continued

Element:	Species	Collection	n	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
ANNELIDA:															
<u>Glycera</u> <u>americana</u>	I	4	1.6(1.2)	14.5(2.2)	4.4(1.7)	4.2(.7)	16.6(5.0)	871(66)	.10(.02)	6.9(1.3)	1.7(.6)	2.9(.6)	5.1(.7)	175(17)	
	II	4	3.3(.5)	15.6(3.1)	5.5(.4)	6.2(2.9)	21.6(1.9)	562(397)	.10(.02)	7.9(2.4)	2.0(.7)	3.9(1.0)	5.1(.7)	224(31)	
	III	3	4.1(.8)	18.3(5.5)	6.6(2.9)	6.7(1.1)	33.3(19.8)	383(286)	.10(.03)	8.0(2.8)	2.5(.3)	4.9(1.0)	5.3(.7)	201(95)	
<u>Glycera</u> <u>robusta</u>	I	4	2.1(1.0)	24.4(8.1)	2.3(1.1)	4.4(1.5)	44.0(10.2)	758(258)	.09(.02)	7.0(2.8)	1.8(1.4)	5.3(1.1)	5.2(.9)	164(4)	
	II	3	3.8(.8)	18.0(5.4)	4.5(1.1)	1.7(1.5)	35.5(24)	629(146)	.10(.01)	7.7(2.4)	1.9(.3)	4.1(1.6)	5.4(.2)	156(30)	
	III	3	3.6(2.0)	49.5(28)	3.0(1.0)	7.1(5.2)	74.0(35)	828(405)	.11(.04)	6.9(4.3)	3.5(2.2)	5.0(1.4)	5.6(.6)	257(28)	
<u>Glycera</u> sp.	I	1	3.7	12.1	8.2	3.0	29.5	699	.13	8.3	5.6	4.0	4.6	192	
ARTHROPODA:															
<u>Callinagaa</u> sp.	I	1	4.4	18.0	10.9	12.3	495	2132	.63	39.0	7.9	6.3	4.3	121	
	III	1	5.3	12.4	9.1	9.5	493	727	.43	22.0	3.9	3.6	6.3	143	
<u>Crabon</u> <u>higromaculata</u>	III	1	1.5	13.5	2.9	8.4	145	101	.30	5.7	1.5	2.4	4.1	104	
	III	1	9.1	8.1	7.3	15.9	234	655	.28	23.0	3.2	8.1	9.8	108	
NEMATEA:															
Unidentified sp.	III	1	1	.9	1	3.0	6.8	175	.07	8.2	1.5	1.6	2.2	187	
ECHINURA															
<u>Listriolobus</u> sp.	I	1	1	7.3	1	234	55	30,827	.51	532	89	35	1.2	171	
	III	1	1	3.7	2.3	181	65	23,696	.39	429	73	30	.8	187	
SIPUNCULA															
Unidentified sp.	II	1	4.0	3.6	2.9	13.6	22.1	1643	.43	33.6	6.1	5.2	7.1	36.8	

Table A-5. EBMUD. Trace element concentrations in parts per million dry weight in *Macoma nasuta* from collections I (27 Nov. -5 Dec. 1974) and II (13-14 Feb. 1975). Three replicates, one standard deviation (), 10 individuals per replicate.

Element	Collection	Stations			
		1	2	3	4
Ag	I	16.1 (6.1)	18.1 (3.1)	21.6 (3.6)	14.2 (2.3)
	II	27.1 ^{a)}	24.0 (1.5)	20.0 (1.0) ^{b)}	13.5 (3.2)
As	I	21.1 (2.9)	20.1 (0.8)	18.9 (1.6)	20.8 (2.1)
	II	18.6 ^{a)}	19.1 (1.6)	29.9 (2.2) ^{b)}	19.2 (1.1)
Cd	I	1.8 (0.7)	1.9 (0.8)	2.6 (0.5)	1.3 (0.4)
	II	1.8 ^{a)}	1.8 (0.4)	1.6 (0.2) ^{b)}	1.0 (0.3)
Cr	I	9.0 (3.2)	5.6 (1.3)	9.7 (2.0)	12.0 (6.6)
	II	18.9 ^{a)}	14.1 (1.0)	20.1 (2.0) ^{b)}	10.1 (1.7)
Cu	I	71.2 (6.1)	65.3 (3.8)	79.2 (10.9)	56.6 (5.9)
	II	82.1 ^{a)}	95.4 (18.6)	149.0 (39.4) ^{b)}	67.9 (27.3)
Fe	I	821 (222)	559 (66)	802 (261)	785 (123)
	II	1387 ^{a)}	1094 (403)	1043 (86) ^{b)}	575 (62)
Hg	I	.67 (.09)	.63 (.04)	.96 (.11)	.68 (.11)
	II	.84 ^{a)}	.76 (.10)	.81 (.04) ^{b)}	.76 (.12)
Mn	I	35.3 (6.7)	19.9 (1.7)	39.5 (8.1)	24.7 (1.9)
	II	77.7 ^{a)}	64.8 (7.3)	101 (17.5) ^{b)}	45.9 (0.9)
Ni	I	8.1 (3.4)	5.9 (1.7)	8.9 (0.5)	8.6 (1.3)
	II	11.1 ^{a)}	10.2 (1.3)	10.8 (1.3) ^{b)}	7.2 (0.4)
Pb	I	8.1 (1.1)	9.1 (1.5)	12.0 (1.0)	11.6 (0.9)
	II	13.2 ^{a)}	12.6 (2.6)	15.3 (0.6) ^{b)}	13.0 (2.6)
Se	I	4.3 (0.3)	4.2 (0.8)	4.2 (0.2)	4.3 (0.4)
	II	5.4 ^{a)}	5.2 (0.1)	7.8 (2.2) ^{b)}	4.0 (0.1)
Zn	I	223 (103)	297 (44)	386 (73)	420 (25)
	II	380 ^{a)}	389 (24)	555 (3) ^{b)}	366 (82)

a) Replicate number equals one.

b) Replicate number equals two.

Table A-6. EBMUD. Trace element concentrations in parts per million dry weight in *Pectinaria californiensis* from collections I (29 Nov.-5 Dec. 1974) and II (13-14 Feb. 1975). One replicate unless otherwise indicated, one standard deviation (), 10-15 individuals per replicate.

Element	Collection	Stations			
		1	2	3	4
Ag	I	2.4	2.7	2.1	2.8
	II	a)	3.5 (0.9) b)	2.9	5.6 (0.1) b)
As	I	9.8	12.2	9.5	8.7
	II	—	12.7 (1.6)	33.4	38.1 (5.2)
Cd	I	2.2	2.4	2.8	1.8
	II	—	2.7 (1.8)	3.6	4.8 (1.3)
Cr	I	11.8	42.2	10.0	12.0
	II	—	22.5 (7.5)	32.9	21.8 (1.3)
Cu	I	21.6	19.8	18.9	17.3
	II	—	28.1 (0.4)	3.4	38.1 (5.2)
Fe %	I	.24	.41	.21	.26
	II	—	.29 (0.11)	.36	.17 (0.04)
Hg	I	.16	.12	.15	.13
	II	—	.21 (0.01)	.25	.25 (0.09)
Mn	I	15.6	19.8	13.2	21.1
	II	—	20.8 (9.2)	25.6	16.9 (5.3)
Ni	I	4.1	7.2	5.4	4.2
	II	—	8.9 (2.8)	9.6	10.8 (1.8)
Pb	I	7.0	6.4	2.0	5.8
	II	—	5.6 (1.2)	6.8	7.2 (2.8)
Se	I	11.3	14.1	10.4	9.8
	II	—	15.0 (1.5)	15.8	10.8 (1.5)
Zn	I	86.2	92.6	81.1	90.0
	II	—	145 (16.4)	223	169 (7.1)

a) station 1, collection II is missing.

b) replicate number is two for stations 2 and 4, collection II.

Table A-7. EBMUD. Trace element concentrations in parts per million dry weight in *Ampelisca milleri* from collections I (5 Dec. 1974) and II (13 Feb. 1975). Three replicates, one standard deviation (), several hundred individuals per replicate.

Station:		1	2	3	4
Element	Collection				
Ag	I	1.7 (.6)	1.8 (.2)	2.5 (.3)	1.9 (.4)
	II	1.0 (.6)	1.8 (.3)	1.9 (.6)	—
As	I	4.6 (.1)	4.9 (.1)	3.3 (.5)	4.4 (.9)
	II	5.6 (.4)	4.4 (.8)	3.3 (.1)	—
Cd	I	<1	<1	1.5 (.2)	2.0 (.2)
	II	<1	<1	1.1 (.6)	—
Cr	I	14.4 (4.4)	6.1 (.8)	7.3 (1.5)	12.8 (1.1)
	II	20.2 (1.3)	19.3 (1.8)	15.3 (1.9)	—
Cu	I	106 (8)	106 (3)	105 (1)	103 (1)
	II	111 (4)	108 (2)	106 (1)	—
Fe	I	1799 (30)	864 (56)	925 (27)	936 (69)
	II	1680 (403)	1522 (65)	1365 (20)	—
Hg	I	.15 (.02)	.15 (.01)	.16 (.01)	.20 (.02)
	II	.19 (.02)	.17 (.02)	.19 (.00)	—
Mn	I	47.3 (1.7)	29.4 (0.8)	33.4 (1.0)	38.1 (3.2)
	II	113 (3)	82.0 (6.7)	68.4 (1.1)	—
Ni	I	4.5 (.5)	2.5 (.3)	2.4 (.1)	4.0 (.3)
	II	6.6 (.1)	6.3 (.1)	4.3 (.3)	—
Pb	I	4.3 (.5)	1.7 (.6)	4.1 (.8)	3.7 (.5)
	II	4.2 (.7)	4.3 (.9)	4.3 (.7)	—
Se	I	2.7 (.4)	2.6 (.2)	2.8 (.3)	3.4 (.6)
	II	3.4 (.1)	3.0 (.3)	3.1 (.1)	—
Zn	I	70.8 (.8)	63.1 (1.5)	61.3 (1.7)	79.2 (1.6)
	II	75.7 (.6)	69.6 (2.3)	67.9 (.9)	—

Table A-8. ERMUD. Trace element concentrations in parts per million dry weight in miscellaneous invertebrates from collections I (25 Nov.-5 Dec. 1974) and II (13-14 Feb. 1975). Means of \bar{n} replicates, one standard deviation (), 5-20 individuals per replicate.

Species	Collection	\bar{n}	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Se	Zn
MOLLUSCA:														
<i>Lyonsia californica</i>	I	4	7.5(.5)	7.8(.5)	10.6(.1)	14.5(25)	90(4.3)	1190 (100)	.66(.03)	92.9(2.9)	11.8(1.2)	22.9(3.7)	11.4(.3)	469(20)
<i>Macoma inquinata</i>	II	3	7.2(2.9)	11.6(4.4)	1.8(.3)	19.4(3.0)	70(18)	1370 (630)	.68(.03)	84.4(32)	8.9(1.4)	12.9(2.7)	6.2(1.6)	404(156)
<i>Musculus senhousia</i>	II	2	20.1(5.6)	8.9(1.3)	3.5(1.6)	14.8(1.4)	31(4.4)	550 (60)	.45(.28)	23.1(14)	4.9(1.7)	7.8(.6)	6.5(.6)	85(3)
<i>Tapes japonica</i>	I	2	4.4(1.5)	6.1(.1)	4.1(.5)	9.3(9.4)	22(.1)	1090 (910)	.26(.02)	17.3(10)	7.1(1.8)	3.9(1.5)	6.9(.9)	125(21)
<i>Protothaca staminea</i>	I	1	4.0	2.2	5.8	5.9	19	350	.35	8.5	2.8	6.0	5.7	81
<i>Tresus nuttallii</i>	I	2	4.3(.1)	7.6(1.6)	3.3(1.2)	31.3(5.9)	22(1)	4500 (510)	.24(.02)	559(304)	9.2(1.3)	11.1(.3)	4.8(.8)	82(.4)
ANNELIDA:														
<i>Asychis elongata</i>	I	3	12.6(1.7)	4.3(1.8)	4.0(1.1)	56.3(.8)	121(22)	8100(4000)	1.06(.19)	117(50)	178(41)	15.7(5.5)	3.3(.8)	84(12)
	II	3	24.0(8.1)	9.0(1.1)	8.2(3.7)	81(24)	139(35)	10100(3200)	2.87(.6)	157(47)	386(136)	20.7(8.1)	4.0(.9)	50(68)
<i>Glycera</i> sp.	I	4	3.3(.5)	15.4(1.5)	3.6(.3)	1.8(.9)	21.4(5.1)	800 (160)	.07(.03)	6.3(.9)	1.6(.9)	4.4(1.2)	4.6(.9)	186(59)
<i>Glycera americana</i>	II	2	2.6(.3)	21.6(8.9)	3.2(.6)	5.1(1.6)	24.6(4.2)	880(280)	.12(.03)	9.5(.6)	2.4(1.1)	4.0(.6)	5.7(1.1)	217(47)
<i>Glycera robusta</i>	II	1	3.6	12.7	1.9	.7	13.9	830	.08	9.3	1.9	6.8	4.8	222
NEMERTEA:														
Unidentified species	II	1	1.5	1.1	1.5	8	16.0	390	.09	38	1.3	1.1	3.8	154

Table A-9. Spoil Site. Trace element concentrations in parts per million dry weight in *Mytilus edulis* transplanted 24-26 Dec. 1975 from collections I (15-16 Jan. 1975), II (26 Jan. 1975), III (6 Feb. 1975), and IV (17 Mar. 1975). Three replicates, one standard deviation (), 10 individuals per replicate.

Stations:			1	2	3	4	5	6
Element	Collection	Depth						
Ag	I	T ¹	1.0(.2)	.8(.6)	—	1.2(1.0)	—	.7(.7)
	II	T	1.5(.3)	.9(.5)	—	1.3(.4)	—	1.3(.2)
	III	T	.8(.6)	1.5(.7)	—	1.7(.4)	—	1.6(.2)
	IV	T	.5(.2)	1.0(.3)	—	1.0(.7)	—	.6(.6)
	I	M ²	.8(.2)	1.0(.4)	.8(.5)	—	1.3(.3)	1.0(.6)
	II	M	1.4(.4)	1.2(.5)	.8(.3)	1.0(.3)	1.3(.4)	1.1(.1)
	III	M	1.4(.3)	.2(.8)	1.1(.2)	.6(.8)	.7(.4)	.8(.4)
	IV	M	1.7(.6)	1.8(.8)	.4(.2)	1.1(.2)	1.9(.4)	1.1(.1)
	I	B ³	1.0(.4)	.8(.2)	.9(.5)	1.0(.3)	.8(.2)	1.3(.1)
	II	B	1.1(.4)	.7(.5)	1.2(.3)	1.6(.4)	1.5(.2)	1.0(.6)
	III	B	.6(.6)	.6(.1)	.8(.3)	1.1(.4)	.6(.2)	1.0(.6)
	IV	B	1.2(.7)	1.6(.4)	.3(.3)	.6(.4)	.7(.6)	.8(.1)
	I	T	7.2(.9)	7.1(1.2)	—	8.6(.6)	—	7.3(1.2)
	II	T	7.0(.9)	7.5(.8)	—	8.2(.5)	—	7.3(1.0)
	III	T	8.2(.7)	9.1(.2)	—	8.7(.13)	—	8.5(.9)
	IV	T	11.4(1.0)	9.1(1.1)	—	10.2(1.0)	—	9.6(.8)
As	I	M	8.2(1.3)	7.0(.9)	7.1(.3)	—	6.4(.4)	7.7(1.1)
	II	M	7.3(.3)	7.0(1.3)	7.5(.7)	6.2(.8)	6.6(1.1)	7.9(.7)
	III	M	8.6(1.1)	8.1(.4)	9.2(.4)	8.4(1.2)	9.4(.7)	8.0(.5)
	IV	M	8.4(.9)	9.0(.1)	8.5(.9)	10.0(1.4)	10.7(2.1)	9.2(.4)
	I	B	6.8(.4)	6.9(.5)	8.0(.3)	7.0(1.0)	7.3(1.4)	7.9(1.0)
	II	B	6.6(1.5)	6.9(.5)	7.0(1.1)	7.1(.9)	6.8(.7)	7.8(.9)
	III	B	8.6(1.5)	10.1(1.3)	8.8(.6)	7.3(.3)	9.3(.9)	10.4(1.2)
	IV	B	9.1(.9)	9.8(.5)	12.2(2.1)	9.3(.7)	10.7(2.2)	8.9(.7)

Table A-9 Continued

Stations:			1	2	3	4	5	6
Element	Collection	Depth						
Cd	I	T	5.3(.7)	5.0(.2)	—	5.8(.9)	—	4.8(.6)
	II	T	5.6(.2)	5.1(.7)	—	5.5(.6)	—	4.8(.2)
	III	T	4.8(.8)	5.2(.7)	—	5.4(.4)	—	5.4(.6)
	IV	T	5.9(.6)	5.9(.5)	—	5.9(.6)	—	5.7(1.4)
	I	M	5.4(.2)	5.2(.7)	5.8(.2)	—	5.3(.8)	5.4(.1)
	II	M	5.4(1.0)	4.4(.4)	6.0(.4)	4.3(.3)	5.4(1.0)	5.7(.5)
	III	M	6.1(.8)	4.4(.5)	5.5(.4)	5.0(1.4)	4.7(.6)	4.0(.6)
	IV	M	6.2(.5)	5.6(.1)	7.6(.5)	5.8(.8)	8.3(1.6)	7.5(.4)
	I	B	6.0(1.2)	5.6(.4)	5.1(.6)	5.1(.3)	4.9(.6)	5.8(.5)
	II	B	5.6(.6)	4.8(.7)	6.2(.5)	4.9(.4)	5.3(.6)	5.3(.7)
	III	B	5.3(.5)	4.7(1.0)	5.7(.6)	4.9(.2)	4.3(.7)	4.7(1.0)
	IV	B	5.8(.4)	5.7(.9)	5.1(.9)	6.0(1.2)	6.7(1.2)	7.7(1.4)
	I	T	2.0(2.2)	.9(1.3)	—	1.7(3.6)	—	2.0(2.2)
	II	T	1	1.7(1.9)	—	.8(1.5)	—	1
	III	T	1	.1(.3)	—	1	—	1.4(1.6)
	IV	T	.7(1.2)	3.0(2.0)	—	4.4(1.6)	—	4.7(1.7)
Cr	I	M	2.8(.4)	.6(1.2)	1.4(2.8)	—	2.2(.9)	.5(2.5)
	II	M	1	1.5(.7)	1	.7(2.4)	2.0(1.3)	2.3(3.6)
	III	M	.3(.7)	1	.9(3.1)	1.4(2.4)	1	.6(.8)
	IV	M	2.2(1.5)	2.0(3.0)	4.0(1.8)	.7(.9)	3.4(1.4)	4.6(2.1)
	I	B	2.3(2.3)	.8(1.3)	2.2(1.4)	2.6(2.9)	1.3(1.1)	2.0(.6)
	II	B	1.3(1.1)	2.4(2.8)	.4(.7)	1.9(1.4)	1.0(.7)	.8(5.9)
	III	B	.3(1.0)	2.3(.5)	2.0(3.3)	1.0(.8)	.6(1.6)	.1(.6)
	IV	B	.5(3.5)	.3(.9)	4.5(1.0)	2.5(1.6)	5.9(2.2)	4.1(1.3)
	I	T	9.1(.4)	9.6(1.8)	—	8.8(.5)	—	10.6(1.5)
	II	T	7.3(.8)	8.3(2.0)	—	10.2(1.0)	—	7.1(1.5)
	III	T	9.6(1.3)	10.1(.7)	—	9.2(1.2)	—	9.2(.5)
	IV	T	12.8(2.3)	9.2(.6)	—	10.0(.4)	—	13.7(.9)
	I	M	8.9(.3)	7.5(.9)	8.3(.7)	—	8.6(.8)	10.7(.1)
	II	M	8.4(1.2)	8.9(1.1)	8.4(.9)	7.5(.4)	7.8(1.8)	9.9(1.7)
	III	M	9.4(1.7)	8.3(1.1)	10.6(2.1)	9.3(2.5)	9.7(.8)	8.0(.8)
	IV	M	10.1(1.9)	8.3(1.1)	9.2(1.1)	11.5(1.3)	15.0(1.7)	11.9(.8)
Cu	I	B	9.4(1.5)	8.3(1.0)	9.8(1.0)	10.6(4.1)	8.9(.5)	10.5(1.1)
	II	B	7.3(1.1)	8.4(.5)	7.4(.9)	9.4(1.5)	7.8(1.2)	9.3(1.0)
	III	B	8.2(1.4)	10.7(.1)	10.1(.4)	9.0(1.5)	9.1(.9)	10.4(1.0)
	IV	B	9.0(1.6)	10.0(1.1)	12.7(1.0)	10.9(1.0)	13.6(2.3)	13.2(1.8)

Table A-9 Continued

Stations:			1	2	3	4	5	6
Element	Collection	Depth						
Fe	I	T	131 (7)	129 (9)	—	154 (25)	—	150 (23)
	II	T	136 (17)	134 (12)	—	139 (29)	—	154 (3)
	III	T	92 (8)	96 (4)	—	118 (28)	—	86 (10)
	IV	T	208 (7)	158 (21)	—	219 (36)	—	200 (42)
	I	M	181 (46)	122 (12)	151 (2)	—	117 (18)	169 (32)
	II	M	120 (31)	130 (6)	148 (4)	102 (10)	160 (18)	148 (7)
	III	M	93 (10)	92 (4)	125 (15)	86 (6)	139 (17)	91 (7)
	IV	M	144 (18)	98 (5)	195 (76)	139 (23)	162 (26)	146 (16)
	I	B	137 (27)	127 (16)	153 (5)	125 (8)	140 (9)	160 (4)
	II	B	139 (26)	158 (16)	169 (32)	135 (14)	174 (55)	225 (57)
	III	B	96 (5)	108 (9)	110 (3)	87 (6)	147 (13)	115 (8)
	IV	B	131 (20)	144 (17)	160 (.4)	164 (13)	317 (47)	172 (18)
Hg	I	T	.33 (.03)	.33 (.02)	—	.35 (.03)	—	.30 (.01)
	II	T	.29 (.03)	.31 (.02)	—	.30 (.02)	—	.32 (.01)
	III	T	.27 (.01)	.32 (.02)	—	.31 (.03)	—	.32 (.03)
	IV	T	.33 (.01)	.29 (.02)	—	.35 (.02)	—	.32 (.06)
	I	M	.37 (.02)	.31 (.03)	.33 (.02)	—	.31 (.02)	.33 (.00)
	II	M	.30 (.02)	.26 (.01)	.31 (.03)	.26 (.04)	.32 (.02)	.30 (.07)
	III	M	.29 (.01)	.24 (.01)	.33 (.03)	.29 (.01)	.34 (.03)	.26 (.01)
	IV	M	.32 (.04)	.29 (.03)	.41 (.04)	.29 (.01)	.38 (.03)	.40 (.05)
	I	B	.44 (.04)	.32 (.02)	.34 (.01)	.32 (.00)	.31 (.04)	.31 (.01)
	II	B	.29 (.01)	.29 (.01)	.33 (.01)	.31 (.02)	.28 (.01)	.31 (.01)
	III	B	.31 (.01)	.30 (.01)	.30 (.01)	.26 (.01)	.39 (.02)	.28 (.02)
	IV	B	.28 (.06)	.37 (.05)	.31 (.04)	.35 (.04)	.40 (.08)	.41 (.04)
Mn	I	T	6.6 (2)	7.0 (1)	—	4.7 (1)	—	8.6 (1)
	II	T	7.4 (2)	6.4 (2)	—	8.7 (4)	—	8.3 (2)
	III	T	8.0 (1)	7.8 (1)	—	7.1 (2)	—	5.4 (2)
	IV	T	13.4 (2)	8.2 (2)	—	9.2 (3)	—	14.4 (1)
	I	M	9.3 (4)	4.8 (1)	8.7 (2)	—	2.7 (1)	7.3 (2)
	II	M	7.3 (1)	6.4 (1)	6.4 (1)	6.1 (1)	6.9 (1)	8.3 (3)
	III	M	6.6 (1)	5.5 (2)	6.5 (2)	5.7 (4)	6.6 (2)	6.0 (1)
	IV	M	9.0 (1)	6.8 (1)	7.7 (4)	12.3 (2)	9.8 (3)	10.4 (2)
	I	B	8.7 (4)	7.0 (4)	8.4 (2)	5.6 (1)	6.3 (1)	11.4 (7)
	II	B	5.7 (2)	6.6 (1)	7.7 (3)	8.1 (1)	8.3 (3)	8.3 (1)
	III	B	5.1 (2)	9.1 (3)	5.4 (2)	5.6 (3)	6.4 (1)	8.9 (3)
	IV	B	5.8 (2)	9.9 (6)	10.8 (1)	9.7 (3)	12.1 (2)	12.9 (2)

Table A-9 Continued

			1	2	3	4	5	6
Element	Collection	Depth						
Ni	I	T	2.3(.5)	1.6(.8)	—	2.0(.5)	—	2.6(.5)
	II	T	1.9(.1)	1.7(.2)	—	1.7(.7)	—	2.3(.5)
	III	T	2.1(1.0)	2.1(.3)	—	2.0(.7)	—	1.7(.3)
	IV	T	3.6(1.1)	1.8(.5)	—	3.4(.8)	—	3.8(1.0)
	I	M	2.4(.2)	1.9(.5)	1.9(.5)	—	1.9(.3)	2.2(.3)
	II	M	1.1(.03)	1.6(.7)	1.3(.5)	.8(.1)	1.7(.7)	1.6(.7)
	III	M	2.2(.3)	1.5(.2)	2.1(.8)	2.2(.6)	2.4(1.0)	1.7(.1)
	IV	M	2.7(.4)	2.1(.1)	3.0(1.4)	3.1(.6)	4.2(.7)	3.4(.9)
	I	B	1.2(.1)	2.2(.7)	2.1(.1)	2.3(.6)	1.4(.8)	2.3(.7)
	II	B	1.9(.6)	1.5(1.0)	2.2(.7)	2.1(.5)	1.7(.2)	3.1(.5)
	III	B	2.4(.3)	2.7(1.1)	2.2(.6)	.8(.9)	2.1(.8)	1.5(.4)
	IV	B	1.8(.6)	3.0(.4)	3.9(.2)	4.1(.6)	4.5(.1)	3.9(.3)
	I	T	3.6(1.2)	3.8(.8)	—	3.4(1.1)	—	3.8(.1)
	II	T	3.8(1.2)	3.0(.4)	—	4.6(2.4)	—	2.5(.03)
	III	T	5.0(1.7)	3.9(.8)	—	4.6(1.4)	—	3.3(1.2)
	IV	T	4.5(1.3)	3.9(.8)	—	3.8(.3)	—	4.8(.6)
Pb	I	M	3.1(.4)	4.1(.7)	3.8(1.0)	—	4.0(.8)	4.0(.6)
	II	M	3.5(1.8)	4.5(1.7)	3.6(.1)	3.8(1.6)	5.0(.5)	3.0(.6)
	III	M	3.7(1.5)	3.1(.6)	4.4(.6)	3.9(.5)	4.6(1.1)	4.8(1.3)
	IV	M	4.1(1.2)	3.1(1.1)	4.2(1.9)	4.3(1.8)	6.0(.6)	3.6(.1)
	I	B	4.8(.1)	4.5(1.1)	3.2(.2)	3.1(.3)	4.2(1.4)	5.6(3.6)
	II	B	3.1(1.6)	5.1(1.5)	3.9(.8)	4.4(.7)	3.4(.4)	4.4(.2)
	III	B	3.5(1.0)	3.5(.3)	4.0(.9)	4.2(.4)	3.7(.9)	3.2(.4)
	IV	B	4.8(.3)	4.0(1.1)	3.7(1.3)	3.7(.9)	4.1(.3)	3.8(1.0)
	I	T	9.2(.5)	9.0(.5)	—	9.6(.4)	—	8.7(1.3)
	II	T	9.4(1.5)	9.0(1.1)	—	9.6(.4)	—	8.7(1.4)
	III	T	9.7(.6)	9.1(.9)	—	10.5(.4)	—	8.9(1.7)
	IV	T	11.3(1.5)	10.4(.7)	—	11.2(.8)	—	10.9(.8)
	I	M	9.6(.5)	8.8(1.0)	7.6(.9)	—	8.7(.7)	9.4(.8)
	II	M	8.6(1.1)	8.9(.5)	9.8(.8)	7.7(.3)	9.7(.6)	9.5(.9)
	III	M	9.7(.8)	7.9(.6)	11.1(.5)	8.8(.5)	9.7(1.1)	8.6(1.1)
	IV	M	9.5(.6)	9.2(.7)	11.6(.5)	10.2(1.2)	11.7(1.7)	11.3(1.1)
Se	I	B	9.0(.3)	8.4(.7)	9.1(.8)	9.3(.7)	8.8(.5)	9.7(.6)
	II	B	8.9(.8)	9.1(.5)	8.8(.4)	9.4(1.0)	8.0(.7)	10.3(1.1)
	III	B	9.6(.7)	10.2(.7)	10.9(1.2)	8.3(.5)	10.2(.7)	10.1(.7)
	IV	B	9.2(.3)	9.6(1.2)	11.3(1.0)	11.1(.4)	10.2(1.0)	10.9(1.3)

Table A-9 Continued

Stations:			1	2	3	4	5	6
Element	Collection	Depth						
Zn	I	T	225(17)	277(12)	—	224(7)	—	199(29)
	II	T	215(53)	225(28)	—	212(16)	—	196(42)
	III	T	243(61)	214(10)	—	218(12)	—	180(15)
	IV	T	311(25)	250(12)	—	277(29)	—	294(72)
	I	M	244(3)	191(21)	204(24)	—	191(18)	222(31)
	II	M	223(15)	166(17)	229(6)	195(14)	211(3)	217(30)
	III	M	227(34)	139(14)	250(17)	189(6)	229(10)	183(16)
	IV	M	233(3)	219(35)	331(19)	264(32)	325(58)	306(22)
	I	B	204(18)	206(35)	204(15)	192(35)	206(18)	241(14)
	II	B	199(29)	186(22)	204(37)	241(71)	194(3)	229(41)
	III	B	208(17)	216(18)	240(6)	213(29)	210(29)	206(23)
	IV	B	253(10)	253(7)	301(10)	290(47)	306(39)	297(9)

1. T=top; 9 meters from bottom.
2. M=middle; 6 meters from bottom.
3. B=bottom; 1 meter from bottom.

a) Transplants at top (9 meters from bottom) missing at stations 1 and 5 in all collections. Transplants at station 4, middle depth were missing in collection I.

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Table A-10. EBMUD. Trace element concentrations in parts per million dry weight in Mytilus edulis transplanted 24-26 Dec. 1974 from collections I (10 Jan. 1975), II (11 Feb. 1975), and III (18 Mar. 1975). Three replicates, one standard deviation (), 10 individuals per replicate.

Stations:			1	2	3	4
Element	Collection	Depth				
Ag	I	T ¹	1.8(.6)	1.4(1.0)	.9(.4)	.8(.3)
	II	T	1.3(.2)	1.0(.4)	1.0(.3)	1.5(.5)
	III	T	1.0(.1)	1.2(.4)	—	1.2(.2)
	I	M ²	1.8(.7)	3.0(.2)	1.7(.5)	1.6(.9)
	II	M	.9(.3)	1.4(.3)	.7(1.0)	.8(.9)
	III	M	1.4(.2)	1.3(.5)	—	1.9(.4)
	I	B ³	1.2(.4)	1.2(1.0)	1.3(.5)	.6(.4)
	II	B	.7(.8)	1.3(.5)	.9(.8)	.8(.1)
	III	B	1.5(.1)	1.1(.2)	—	.8(.5)
As	I	T	5.6(1.5)	5.4(.5)	5.3(.9)	5.6(.5)
	II	T	7.5(.5)	9.0(1.6)	7.8(1.9)	7.4(.7)
	III	T	7.9(1.5)	8.2(1.1)	—	8.4(1.2)
	I	M	4.1(1.1)	6.2(.3)	6.0(1.0)	6.1(.7)
	II	M	7.2(.5)	8.2(1.5)	8.1(1.0)	7.5(1.0)
	III	M	8.3(.6)	8.3(.9)	—	5.8(.7)
	I	B	5.6(.5)	5.8(1.0)	5.2(1.1)	6.0(.4)
	II	B	6.8(.5)	7.2(.4)	8.1(.8)	7.3(.8)
	III	B	7.3(.5)	7.9(1.0)	—	8.7(.7)
Cd	I	T	5.8(.5)	5.7(.6)	5.3(.5)	4.8(.9)
	II	T	5.2(1.0)	5.7(.6)	5.0(.4)	5.9(1.0)
	III	T	6.4(.5)	6.4(.5)	—	6.0(.5)
	I	M	5.9(1.1)	7.5(.5)	6.0(.5)	4.8(.4)
	II	M	4.5(.4)	5.5(.3)	4.7(.9)	5.3(.4)
	III	M	6.8(.4)	6.9(.6)	—	7.1(1.1)
	I	B	4.9(.4)	5.0(.4)	4.9(.8)	4.5(.5)
	II	B	5.0(.5)	5.2(.5)	5.3(.7)	4.5(.5)
	III	B	7.8(.2)	6.5(.3)	—	6.7(1.5)

Tabl A-10 Continued

Stations:			1	2	3	4
Element	Collection	Depth				
Cr	I	T	2.5(2.0)	1.6(1.3)	2.0(1.0)	2.2(2.4)
	II	T	4.5(.6)	1	1.8(1.9)	1.6(.6)
	III	T	3.6(.8)	4.6(1.8)	—	.7(2.1)
	I	M	.8(1.7)	2.8(1.7)	1.8(3.0)	2.5(2.8)
	II	M	.8(2.3)	1.6(3.2)	.8(1.7)	3.1(.7)
	III	M	2.5(1.5)	2.9(1.4)	—	4.1(.5)
	I	B	.5(.8)	.3(2.0)	2.5(2.0)	1.2(3.3)
	II	B	2.9(3.9)	1.5(2.7)	2.5(2.1)	2.2(1.1)
	III	B	1.8(.8)	1.2(3.3)	—	4.5(1.7)
Cu	I	T	7.6(.5)	12.2(8.7)	7.4(.3)	7.9(1.0)
	II	T	9.8(.4)	11.0(1.7)	10.9(.8)	9.7(.8)
	III	T	10.3(2.0)	11.7(1.3)	—	16.9(9)
	I	M	9.0(1.5)	9.0(1.9)	10.6(1.6)	9.2(1.0)
	II	M	9.5(1.1)	9.7(1.1)	11.0(1.1)	9.9(1.1)
	III	M	11.7(1.9)	11.3(.4)	—	11.6(.8)
	I	B	8.4(.1)	7.0(.8)	8.2(.6)	8.3(.8)
	II	B	8.4(.5)	11.6(.7)	10.2(.4)	9.5(1.0)
	III	B	14.0(4.5)	10.6(1.3)	—	12.0(.8)
Fe	I	T	115(6)	96(8)	145(37)	111(21)
	II	T	113(13)	105(13)	108(10)	91(14)
	III	T	152(7)	138(1.6)	—	143(19)
	I	M	119(12)	106(8)	120(13)	294(284)
	II	M	92(5)	134(15)	117(6)	134(8)
	III	M	127(13)	159(15)	—	139(9)
	I	B	117(17)	114(16)	117(4)	97(6)
	II	B	140(9)	130(21)	104(10)	114(2)
	III	B	174(28)	199(11)	—	119(15)

Table A-10 Continued

Stations:			1	2	3	4
Element	Collection	Depth				
Hg	I	T	.33(.04)	.40(.10)	.33(.02)	.33(.05)
	II	T	.39(.02)	.43(.04)	.45(.04)	.43(.07)
	III	T	.51(.04)	.50(.05)	—	.50(.01)
	I	M	.36(.02)	.36(.02)	.35(.04)	.33(.02)
	II	M	.35(.03)	.41(.02)	.38(.01)	.47(.03)
	III	M	.57(.05)	.58(.02)	—	.60(.05)
	I	B	.36(.02)	.37(.02)	.35(.02)	.34(.01)
	II	B	.37(.01)	.42(.02)	.51(.06)	.43(.02)
	III	B	.58(.02)	.47(.05)	—	.53(.03)
Mn	I	T	6.1(.8)	6.8(1.1)	8.9(2.1)	5.8(2.7)
	II	T	10.2(5)	9.6(2.5)	6.6(1.3)	5.8(2.3)
	III	T	10.3(.8)	7.3(.8)	—	12.1(6.7)
	I	M	5.5(2.3)	7.0(1.0)	8.4(2.6)	8.2(4.8)
	II	M	6.0(2.0)	6.8(.8)	7.9(2.5)	8.8(3.0)
	III	M	7.3(2.5)	9.4(2.2)	—	15.0(11)
	I	B	5.4(1.7)	7.4(1.7)	8.0(1.9)	5.3(3.2)
	II	B	5.2(1.4)	8.9(1.4)	16.1(15)	5.8(3.4)
	III	B	9.0(1.4)	11.3(2.2)	—	7.3(.4)
Ni	I	T	1.3(.3)	1.9(.7)	1.7(.3)	2.0(.6)
	II	T	2.6(.3)	2.9(.1)	2.6(.7)	2.1(.7)
	III	T	2.7(.4)	3.3(1.1)	—	3.9(1.6)
	I	M	.9(.8)	1.5(.7)	2.3(.6)	2.9(2.0)
	II	M	1.5(.5)	2.9(1.0)	2.2(.4)	2.6(.6)
	III	M	4.0(.5)	3.4(.7)	—	3.7(.2)
	I	B	1.6(.2)	1.8(.6)	2.0(.7)	1.0(.3)
	II	B	2.5(.6)	3.2(.1)	2.6(.4)	2.2(.9)
	III	B	4.9(2.0)	3.2(.9)	—	3.7(.3)

Table A-10 Continued

Stations:			1	2	3	4
Element	Collection	Depth				
Pb	I	T	3.7(.9)	3.7(.8)	4.3(.6)	4.0(.9)
	II	T	4.0(.2)	3.8(.9)	5.1(.9)	4.2(1.3)
	III	T	6.1(.4)	8.1(1.0)	—	5.7(1.9)
	I	M	3.4(.6)	5.7(4.3)	5.0(.3)	5.4(2.4)
	II	M	5.0(.8)	5.0(.6)	4.7(.9)	4.5(.6)
	III	M	7.7(1.3)	6.9(1.1)	—	8.3(1.4)
	I	B	4.6(1.5)	4.7(1.4)	4.8(.1)	3.2(.2)
	II	B	5.7(1.6)	5.1(.7)	3.7(.8)	5.7(.5)
	III	B	7.1(2.2)	6.1(.8)	—	5.4(.6)
Se	I	T	6.3(2)	6.3(.8)	6.3(.3)	6.7(.8)
	II	T	7.5(.4)	7.7(.6)	7.4(.8)	7.2(.2)
	III	T	7.9(1.1)	8.3(.8)	—	9.2(2.3)
	I	M	5.5(4)	7.2(.4)	6.7(.8)	7.0(.2)
	II	M	8.0(.3)	7.8(.9)	7.8(.4)	6.9(.6)
	III	M	9.5(.8)	9.7(1.0)	—	7.0(.3)
	I	B	6.2(.3)	6.9(.6)	6.3(.7)	7.3(.3)
	II	B	7.6(.7)	8.8(1.0)	7.6(.6)	7.8(.3)
	III	B	9.6(1.1)	7.1(.9)	—	8.4(.8)
Zn	I	T	211(5)	217(12)	227(19)	242(50)
	II	T	291(35)	295(16)	270(37)	282(30)
	III	T	389(66)	454(66)	—	314(58)
	I	M	179(38)	224(26)	240(70)	217(18)
	II	M	253(18)	264(26)	278(47)	266(39)
	III	M	419(27)	375(72)	—	349(42)
	I	B	170(25)	215(31)	230(34)	209(14)
	II	B	238(21)	319(12)	309(39)	282(37)
	III	B	411(17)	339(57)	—	380(62)

1. T=top; 9 meters from bottom.
 2. M=middle; 6 meters from bottom.
 3. B=bottom; 1 meter from bottom.

Table A-11. Berkeley Pier. Trace element concentrations in parts per million dry weight in *Mytilus edulis* controls from collections I (29 Nov. 1974), II (26 Dec. 1974), III (27 Feb. 1975), and (4 Mar. 1975). Three replicates, one standard deviation (), 10 individuals per replicate, shell length, 25-40 mm.

Collection:	I		II		III		IV	
Element								
Ag	1.4	(.4)	1.2	(.4)	1.6	(.2)	1.5	(.8)
As	3.6	(1.0)	5.9	(1.2)	7.6	(.6)	6.1	(1.1)
Cd	4.2	(.01)	4.6	(.1)	6.8	(1.0)	7.2	(.5)
Cr	.6	(2.5)	2.5	(.9)	1.1	(2.2)	2.1	(3.0)
Cu	5.6	(1.2)	7.1	(.1)	8.8	(.4)	10.6	(.5)
Fe	83	(9)	91	(5)	133	(25)	134	(6)
Hg	.25	(.01)	.23	(.03)	.49	(.04)	.52	(.01)
Mn	4.3	(2.0)	6.4	(1.8)	10.0	(2.4)	9.5	(1.9)
Ni	.3	(.5)	1.6	(.3)	3.0	(.7)	3.6	(.4)
Pb	3.2	(1.7)	2.9	(.7)	4.8	(.9)	5.6	(.4)
Se	5.9	(.2)	7.3	(.7)	9.9	(.4)	9.4	(.9)
Zn	127	(6)	206	(58)	245	(88)	275	(21)

Table A-12. Spoil Site. Chlorinated hydrocarbon concentrations in parts per billion wet weight in Mytilus edulis transplanted 24-26 Dec. 1974. One replicate, 10 individuals per replicate.

Collection	Station	Level	P,P'- DDE	P,P'- DDD	PCB Aroclor 1254
I (16 Jan. 1975)	1	top	7.5	4.0	77
		mid	5.3	4.7	93
		bot	5.3	2.8	55
	2	top	4.7	2.6	44
		mid	5.9	5.5	121
		bot	6.2	2.4	53
	3	top ^{a)}	-	-	-
		mid	5.4	6.1	128
		bot	5.2	3.2	52
	4	top ^{a)}	5.2	2.8	51
		mid ^{a)}	-	-	-
		bot	5.2	2.5	41
	5	top ^{a)}	-	-	-
		mid	3.8	5.1	107
		bot	5.2	2.8	47
	6	top	7.6	4.1	66
		mid	5.1	4.0	91
		bot	5.4	3.1	52
Mean			5.5 (1.0)	3.7 (1.2)	72 (29)
II (26 Jan. 1975)	1	top	7.8	3.4	66
		mid	5.3	3.5	63
		bot	9.2	4.7	80
	2	top	5.9	2.9	47
		mid	4.4	4.2	84
		bot	4.4	2.1	36
	3	top ^{a)}	-	-	-
		mid	7.2	4.6	98
		bot	4.1	2.2	31
	4	top	7.3	2.9	45
		mid	5.2	5.7	84
		bot	4.9	2.4	35
	5	top ^{a)}	-	-	-
		mid ^{a)}	5.4	4.1	64
		bot ^{a)}	-	-	-
	6	top	7.2	2.8	45
		mid	4.4	4.4	90
		bot	5.6	3.3	50
Mean			4.8 (1.8)	8.5 (1.0)	61 (22)

Table A-12 Continued

Collection	Station	Level	p,p'- DDE	p,p'- DDD	PCB Aroclor 1254	
III (6 Feb. 1975)	1	top	5.0	4.0	74	
		mid	2.3	2.1	49	
		bot	2.7	4.9	83	
	2	top	4.5	4.9	65	
		mid	5.1	4.4	83	
		bot	4.3	3.6	71	
	3	top ^{a)}	-	-	-	
		mid	4.5	4.5	75	
		bot	1.7	2.4	50	
	4	top	7.1	6.3	106	
		mid	4.6	3.8	70	
		bot	3.8	3.9	65	
	5	top ^{a)}	-	-	-	
		mid	2.5	3.1	64	
		bot	2.0	2.6	49	
	6	top	6.4	6.3	77	
		mid	3.7	3.2	69	
		bot	4.7	6.4	86	
	Mean			4.0 (1.5)	4.1 (1.4)	71 (15)
	IV (17 Mar. 1975)	1	top	4.1	3.3	106
			mid	3.0	1.8	103
			bot	2.1	1.6	81
		2	top	2.0	2.2	84
			mid	2.0	1.7	76
bot			1.2	3.5	52	
3		top ^{a)}	-	-	-	
		mid	1.4	2.5	34	
		bot	1.1	1.5	31	
4		top	4.0	3.7	81	
		mid ^{a)}	3.0	4.1	67	
		bot	-	-	-	
5		top ^{a)}	-	-	-	
		mid	1.7	3.4	72	
		bot	1.1	0.8	63	
6		top ^{a)}	3.6	2.2	56	
		mid	-	-	-	
		bot	4.5	3.6	74	
Mean			2.5 (1.2)	2.6 (1.0)	70 (23)	

a) sample missing

Table A-13. EBMUD. Chlorinated hydrocarbon concentrations in parts per billion wet weight in Mytilus edulis transplanted 24-26 Dec. 1974.

Collection	Station	Level	P,P'- DDE	P,P'- DDD	PCB Aroclor 1254
I (10 Jan. 1975)	1	top	5.8	3.3	78
		mid	7.2	3.3	81
		bot	5.7	3.8	71
	2	top	5.2	3.6	66
		mid	6.8	4.3	81
		bot	7.6	3.2	86
	3	top	7.3	4.9	107
		mid	8.1	4.1	78
		bot	6.6	3.1	71
	4	top	4.7	2.7	64
		mid	5.3	3.5	71
		bot	4.7	3.6	67
Mean			6.2 (1.2)	3.6 (0.6)	77 (12)
II (11 Feb. 1975)	1	top	4.4	8.7	121
		mid	7.3	4.8	68
		bot	2.1	5.1	76
	2	top	5.2	8.7	101
		mid	0.9	5.7	80
		bot	3.7	5.2	77
	3	top	3.9	8.3	107
		mid	1.7	4.5	79
		bot	1.1	2.6	46
	4	top	4.3	8.7	79
		mid	1.9	5.4	88
		bot	1.9	4.6	77
Mean			3.1 (1.9)	6.0 (2.1)	83 (19)
III (18 Mar. 1975)	1	top	1.0	11.1	127
		mid	2.1	a)	126
		bot	1.6	a)	193
	2	top	0.6	3.9	157
		mid	0.5	9.7	149
		bot	3.5	9.7	193
	4	top	1.7	9.7	130
		mid	2.1	6.4	133
		bot	1.3	7.8	115
Mean			1.6 (0.9)	8.3 (2.5)	147 (29)

a) Substances present interfering with DDT detection.

Table A-14. Spoil Site. Percent of sand, silt, and clays in sediments at 4 depths from collection I (20-26 Dec. 1974). Three replicates unless otherwise indicated, one standard deviation ().

		Depth (cm)			
		0-2	2-10	10-30	30-60
Sand	Stations				
	1	41.2 (8.0)	43.6 (6.0)	43.3 (4.4)	36.5 (2.9)
	5	46.2 (2.5)	49.5 (1.7)	48.6 (6.7)	43.1 (0.8)
	6	31.4 (2.7)	32.9 (3.6)	34.7 (2.8)	35.2 (0.6) a)
Silt	1	34.2 (3.8)	33.9 (2.8)	32.5 (2.4)	35.6 (1.7)
	5	35.1 (2.1)	33.0 (1.9)	32.4 (6.5)	34.0 (1.1)
	6	43.8 (2.0)	43.8 (2.7)	43.0 (2.8)	41.3 (0.6) a)
Coarse Clay	1	9.4 (1.7)	9.1 (1.7)	9.1 (0.8)	17.8 (6.9)
	5	6.3 (0.4)	6.1 (0.5)	6.5 (1.2)	11.8 (7.8)
	6	7.9 (0.4)	7.5 (0.4)	6.9 (0.8)	6.9 (0.8) a)
Fine Clay	1	21.3 (9.5)	13.4 (1.8)	15.1 (1.4)	10.1 (8.1)
	5	12.4 (0.5)	11.4 (0.8)	12.5 (1.2)	11.1 (6.5)
	6	16.9 (0.9)	15.8 (0.8)	15.4 (0.7)	16.6 (0.7) a)

a) Replicate number is two

Table A-15. Spoil Site. Percent of sand, silt, and clays in surface (0-2 cm) sediments from collections I, II, and III and from the impact area center, in sediments from the Oakland Inner Harbor dredge site, and in sediments from the dredge hoppers. Three replicates, unless otherwise indicated, one standard deviation ().

		2 Jan 75 I	26 Jan 75 II	Oak.In.Har. Dredge Site Hopper Samp.	19-20 Feb 75 III	Impact Area Center ^{a,b)}
	<u>Stations</u>					
Sand	1	41.2 (8.0)	49.4 (2.6)		34.4 (2.1)	
	2	49.6 (3.9)	43.6 (2.1)		34.7 (2.3)	
	3	18.8 (11.7)	20.3 (0.6)	45.9 (13.7) 33.5 (24.6)	10.6 (2.1)	64.4
	4	33.9 (2.6)	24.6 (2.3)		25.4 (1.8)	
	5	46.2 (2.5)	31.3 a)		34.1 (2.0)	
	6	31.4 (2.7)	32.1 (2.8)		29.8 (1.5)	
Silt	1	34.2 (3.8)	33.2 (2.6)		40.7 (1.6)	
	2	31.1 (3.2)	37.3 (2.1)		40.2 (1.0)	
	3	44.5 (6.6)	50.5 (1.7)	25.3 (6.7) 23.4 (8.8)	52.4 (1.6)	17.3
	4	41.9 (0.7)	48.4 (3.5)		46.2 (1.7)	
	5	35.1 (2.1)	41.0 a)		42.2 (1.1)	
	6	43.8 (2.0)	39.5 (3.1)		43.4 (1.7)	
Coarse Clay	1	9.4 (1.7)	5.2 (1.3)		8.4 (1.1)	
	2	6.3 (0.4)	6.6 (1.3)		8.6 (0.8)	
	3	9.1 (0.4)	8.5 (1.8)	8.9 (2.2) 13.1 (4.2)	11.8 (1.0)	5.0
	4	7.9 (0.9)	7.3 (4.3)		9.4 (0.6)	
	5	6.3 (0.4)	8.9 a)		8.7 (0.2)	
	6	7.9 (0.4)	11.0 (2.6)		9.2 (0.2)	
Fine Clay	1	21.3 (9.5)	12.2 (0.9)		16.6 (0.6)	
	2	13.1 (1.1)	12.4 (0.3)		16.5 (1.1)	
	3	18.8 (0.6)	20.7 (0.7)	20.0 (4.9) 29.5 (11.3)	25.2 (1.4)	9.1
	4	16.3 (1.9)	19.7 (1.3)		19.0 (0.8)	
	5	12.4 (0.5)	18.8 a)		15.0 (0.8)	
	6	16.9 (0.9)	17.4 (1.0)		17.7 (0.3)	4.3 ^{c)}

a) Replicate number equals one.

b) Single sample collected 20 Feb. 1975.

c) Gravel

Table A-16. Spoil Site. Percent of sand, silt, and clays in settled particulates from settling tubes. Three replicates unless otherwise indicated, one standard deviation ().

		29 Jan 75	8 Feb 75	17 Mar 75
		I	II	III
	<u>Stations</u>			
Sand	1	6.9 (0.7)	7.7 (0.8)	18.2 (3.4)
	2	8.6 (1.3)	26.9 (6.0)	17.6 (2.6)
	3	2.3 ^{a)}	2.1 (1.2)	7.3 (0.5)
	4	6.9 ^{a)}	4.5 ^{a)}	12.7 (1.0)
	5	6.3 (4.4)	4.1 (0.8)	17.4 (5.8)
	6	9.3 (1.0)	b)	b)
Silt	1	45.4 (0.7)	45.2 (1.1)	48.7 (0.9)
	2	45.1 (1.1)	37.8 (3.6)	48.4 (1.8)
	3	46.1 ^{a)}	49.4 (15.1)	54.0 (0.6)
	4	51.3 ^{a)}	47.8 ^{a)}	44.3 (0.9)
	5	46.9 (0.7)	47.4 (1.4)	48.6 (2.8)
	6	47.2 (0.3)	b)	b)
Coarse Clay	1	15.3 (0.2)	14.5 (0.8)	10.7 (1.5)
	2	15.0 (0.5)	11.6 (1.1)	11.3 (0.4)
	3	16.2 ^{a)}	15.8 (5.2)	12.5 (0.2)
	4	16.3 ^{a)}	13.8 ^{a)}	16.2 (0.4)
	5	13.9 (1.2)	14.1 (1.2)	11.1 (1.2)
	6	14.5 (0.1)	b)	b)
Fine Clay	1	32.4 (1.1)	33.0 (0.8)	22.2 (2.7)
	2	31.3 (1.2)	23.8 (1.3)	22.8 (0.4)
	3	35.5 ^{a)}	32.6 (9.5)	26.3 (0.4)
	4	25.6 ^{a)}	33.9 ^{a)}	26.8 (2.0)
	5	29.9 (1.2)	34.5 (2.3)	23.0 (1.7)
	6	29.0 (1.2)	b)	b)

a) replicate number is one.
b) missing settling tubes.

Table A-17. Spoil Site. Rate of deposition ($\text{mg}/\text{cm}^2/\text{day}$) of settled particulates and residence times (RT) of settling tubes in days. Three replicates unless otherwise indicated, one standard deviation.

Stations	29 Jan 75		8 Feb 75		17 Mar 75	
	RT	$\text{mg}/\text{cm}^2/\text{day}$ I	RT	$\text{mg}/\text{cm}^2/\text{day}$ II	RT	$\text{mg}/\text{cm}^2/\text{day}$ III
1	27	120 (17)	10	148 (3)	37	158 (14)
2	27	86 (4)	10	282 (15)	37	142 (16)
3	27	63 ^{a)}	10	162 (64)	37	117 (19)
4	34	86 ^{a)}	12	120 ^{a)}	35	93 (6)
5	34	105 (3)	12	122 (11)	36	130 (6)
6	34	103 (6)	b)	—	b)	—

a) replicate number is one.

b) missing settling tubes.

Table A-18. EBMUD. Percent of sand, silt, and clays in sediments at 4 depths from collection I (13-14 Dec. 1974). Three replicates, one standard deviation ().

		Depth (cm)			
		0-2	2-10	10-30	30-60
<u>Stations</u>					
Sand	1	4.0 (0.5)	3.1 (0.2)	30.2 (4.3)	27.4 (2.6)
	2	5.2 (0.3)	4.4 (1.2)	15.7 (10.2)	42.7 (5.6)
	3	5.6 (3.2)	2.8 (1.4)	6.9 (4.1)	20.0 (4.9)
	4	20.2 (3.4)	23.4 (2.8)	44.6 (11.5)	39.3 (6.9)
Silt	1	40.7 (1.3)	41.9 (0.4)	30.7 (3.4)	34.4 (0.2)
	2	40.5 (0.8)	39.8 (1.4)	37.6 (3.7)	23.6 (2.3)
	3	39.4 (1.3)	40.3 (1.4)	38.7 (0.2)	36.0 (1.4)
	4	36.0 (1.4)	34.3 (1.5)	26.0 (2.2)	30.7 (6.0)
Coarse Clay	1	18.1 (0.7)	17.8 (0.5)	13.1 (1.4)	13.3 (1.1)
	2	18.2 (0.5)	18.8 (0.6)	15.7 (3.4)	9.3 (1.1)
	3	18.3 (1.9)	18.7 (1.5)	18.3 (1.6)	14.8 (0.9)
	4	13.9 (0.6)	13.3 (0.3)	8.8 (3.6)	8.3 (3.5)
Fine Clay	1	37.6 (1.6)	37.2 (0.3)	26.1 (3.3)	25.0 (1.3)
	2	36.1 (0.2)	37.1 (0.8)	31.0 (3.4)	17.5 (2.7)
	3	36.7 (2.5)	37.7 (1.4)	36.2 (2.5)	28.8 (1.7)
	4	29.9 (1.5)	28.6 (1.4)	20.6 (3.6)	21.7 (3.0)

Table A-19. EBMUD. Percent of sand, silt, and clays in surface sediments (0-2 cm) from collections I (13 Dec. 1974) and II (17 Feb. 1975). Three replicates, one standard deviation ().

Collection		I	II
<u>Stations</u>			
Sand	1	4.0 (0.5)	2.1 (0.5)
	2	5.2 (0.3)	2.4 (0.8)
	3	5.6 (3.2)	9.8 (1.8)
	4	20.2 (3.4)	14.1 (6.0)
Silt	1	40.7 (1.3)	40.2 (0.2)
	2	40.5 (0.8)	38.5 (2.0)
	3	39.4 (1.3)	39.5 (3.2)
	4	36.0 (1.4)	35.3 (3.3)
Coarse Clay	1	18.1 (0.7)	19.0 (0.2)
	2	18.2 (0.5)	19.2 (2.6)
	3	18.3 (1.9)	14.7 (0.3)
	4	13.9 (0.6)	16.0 (1.8)
Fine Clay	1	37.6 (1.6)	38.7 (0.2)
	2	36.1 (0.2)	39.9 (1.1)
	3	36.7 (2.5)	36.1 (5.4)
	4	29.9 (1.5)	34.6 (3.0)

Table A-20. EBMUD. Rates of deposition ($\text{mg}/\text{cm}^2/\text{day}$) of settled particulates and residence times (RT) of settling tubes in days. Three replicates per sample, one standard deviation ().

Stations	12-14 Jan 75		11-17 Feb 75		18 Mar 75	
	I		II		III	
	RT	$\text{mg}/\text{cm}^2/\text{day}$	RT	$\text{mg}/\text{cm}^2/\text{day}$	RT	$\text{mg}/\text{cm}^2/\text{day}$
1	28	87 (9)	32	69 (1)	35	63 (10)
2	28	85 (8)	32	88 (16)	35	94 (10)
3	a)	a)	31	74 (1)	a)	a)
4	32	87 (10)	33	79 (7)	29	104 (15)

a) missing settling tubes

Table A-21. EBMUD. Percent of sand, silt, and clays in settled particulates from settling tubes. Three replicates, one standard deviation ().

		10-14 Jan. 75	11-17 Feb. 75	18 Mar. 75
		I	II	III
	<u>Stations</u>			
Sand	1	0.5 (.06)	0.3 (.06)	0.6 (0.1)
	2	0.4 (.01)	0.2 (.09)	1.0 (0.5)
	3	a)	0.2 (.01)	a)
	4	0.9 (0.1)	0.6 (.03)	0.8 (.01)
Silt	1	45.8 (5.3)	39.5 (1.3)	37.9 (6.2)
	2	44.1 (7.7)	38.6 (2.3)	41.1 (1.6)
	3	a)	37.5 (7.1)	a)
	4	39.3 (0.3)	39.3 (3.6)	42.3 (1.1)
Coarse Clay	1	18.4 (1.4)	20.1 (1.2)	20.9 (5.1)
	2	20.6 (4.5)	19.7 (0.9)	18.8 (0.5)
	3	a)	20.4 (2.1)	a)
	4	19.8 (0.8)	18.9 (0.9)	17.6 (0.8)
Fine Clay	1	35.4 (3.9)	40.1 (1.3)	40.5 (1.9)
	2	34.9 (3.2)	41.4 (2.0)	39.2 (0.6)
	3	a)	41.9 (4.9)	a)
	4	40.0 (0.7)	41.2 (2.8)	39.3 (0.3)

a) missing settling tubes

Table A-22. Spoil Site. Trace element concentrations in parts per million dry weight (except Fe) in sediments at 4 depths from collection I (26 Dec. 1974 - 2 Jan. 1975). Three replicates, one standard deviation ().

Elements	Depth (cm)	Stations					
		1	2	3	4	5	6
Ag	0-2	1.2 (0.7)	0.4 (0.5)	1.8 (1.0)	0.9 (0.9)	0.5 (0.6)	0.3 (1.1)
	2-10	1.1 (0.5)	1.0 (1.2)	2.8 (0.9)	1.5 (0.2)	0.7 (0.8)	1.7 (0.3)
	10-30	0.9 (1.5)	1.5 (0.5)	0.5 (1.8)	1.8 (1.6)	2.7 (1.5)	1.2 (0.2)
	30-60	1.2 (0.5)	1.1 (1.9)	1.6 (0.6)	2.0 (0.5)	3.2 (0.7)	0.9 (0.9)
As	0-2	6.3 (3.3)	8.8 (2.8)	7.7 (2.8)	8.3 (1.6)	5.1 (1.9)	7.7 (5.0)
	2-10	6.3 (2.6)	5.7 (1.2)	5.5 (0.9)	5.5 (2.2)	5.9 (2.3)	7.2 (0.9)
	10-30	9.2 (3.2)	6.0 (4.0)	7.6 (8.0)	7.0 (0.7)	7.2 (2.6)	5.6 (1.4)
	30-60	5.3 (5.0)	9.2 (2.5)	9.2 (0.8)	11.7 (2.1)	7.9 (0.1)	12.3 (4.1)
Cd	0-2	2.4 (0.8)	-0.3 (1.1)	1.7 (0.1)	1.9 (1.2)	0.2 (0.7)	-0.9 (2.4)
	2-10	1.4 (1.4)	0.6 (1.2)	1.8 (1.1)	2.1 (0.5)	1.1 (1.6)	1.6 (0.2)
	10-30	0.9 (1.3)	1.6 (1.2)	-0.1 (0.4)	1.3 (1.3)	2.2 (0.8)	0.9 (0.2)
	30-60	1.2 (0.6)	0.5 (1.3)	0.6 (0.5)	2.2 (1.1)	3.2 (0.7)	0.8 (0.8)
Cr	0-2	163 (12)	208 (81)	195 (46)	199 (23)	187 (25)	241 (24)
	2-10	197 (4)	179 (41)	219 (34)	195 (4)	226 (35)	198 (2)
	10-30	200 (41)	186 (32)	185 (128)	201 (17)	156 (11)	219 (16)
	30-60	180 (23)	218 (10)	239 (24)	250 (31)	203 (7)	261 (44)
Cu	0-2	32 (4)	50 (28)	50 (10)	41 (3)	41 (12)	51 (3)
	2-10	35 (3)	31 (1)	47 (4)	41 (2)	36 (5)	44 (4)
	10-30	33 (3)	32 (5)	44 (34)	43 (5)	39 (2)	43 (2)
	30-60	36 (7)	73 (41)	52 (5)	50 (5)	38 (2)	54 (14)
Fe %	0-2	2.8 (.4)	2.6 (.5)	3.6 (.6)	3.4 (.1)	3.0 (.2)	4.0 (.2)
	2-10	3.1 (.2)	2.7 (.1)	3.6 (.1)	3.2 (.2)	3.1 (.2)	3.4 (.4)
	10-30	3.0 (.4)	2.7 (.2)	3.0 (2.4)	3.4 (.2)	3.0 (.03)	3.3 (.2)
	30-60	3.0 (.3)	3.6 (.3)	3.8 (.3)	3.7 (.1)	3.0 (.04)	3.9 (.4)

Table A-22 Continued

Elements	Depth (cm)	Stations					
		1	2	3	4	5	6
Hg	0-2	.24 (.02)	.27 (.05)	.35 (.05)	.24 (.01)	.22 (.03)	.30 (.05)
	2-10	.21 (.02)	.31 (.07)	.29 (.03)	.26 (.01)	.22 (.04)	.28 (.03)
	10-30	.32 (.06)	.36 (.01)	.35 (.07)	.29 (.08)	.27 (.01)	.27 (.03)
	30-60	.37 (.05)	.39 (.06)	.40 (.09)	.33 (.01)	.30 (.03)	.32 (.03)
Mn	0-2	382 (25)	333 (33)	460 (98)	410 (27)	399 (44)	533 (47)
	2-10	391 (33)	314 (22)	449 (35)	374 (11)	387 (20)	471 (159)
	10-30	351 (52)	307 (21)	365 (28)	421 (30)	361 (20)	412 (38)
	30-60	341 (48)	418 (35)	455 (25)	421 (33)	335 (1)	505 (110)
Ni	0-2	89 (14)	96 (10)	119 (31)	107 (6)	88 (4)	126 (12)
	2-10	100 (6)	93 (9)	117 (6)	96 (7)	94 (12)	113 (8)
	10-30	98 (19)	88 (7)	99 (66)	102 (4)	92 (3)	106 (10)
	10-30	95 (17)	121 (13)	129 (8)	110 (4)	91 (4)	123 (14)
Pb	0-2	29 (7)	29 (2)	39 (6)	31 (6)	34 (.4)	42 (9)
	2-10	32 (12)	26 (3)	40 (1)	39 (3)	33 (8)	36 (4)
	10-30	23 (4)	29 (5)	37 (23)	35 (6)	32 (.3)	37 (1)
	30-60	31 (5)	37 (2)	39 (2)	39 (5)	32 (2)	43 (13)
Se	0-2	1.0 (0.5)	-0.8 (3.8)	1.9 (1.0)	2.4 (0.9)	1.8 (1.1)	3.4 (0.6)
	2-10	1.8 (0.3)	1.2 (1.2)	1.2 (1.3)	1.6 (1.0)	2.0 (1.8)	2.7 (0.4)
	10-30	1.4 (0.5)	1.3 (1.2)	1.8 (1.6)	1.8 (1.0)	1.9 (1.0)	2.5 (1.0)
	30-60	0.8 (0.8)	1.0 (1.2)	1.5 (0.9)	1.2 (0.5)	0.7 (0.8)	2.0 (1.2)
Zn	0-2	88 (11)	97 (16)	120 (25)	105 (4)	99 (6)	132 (9)
	2-10	97 (5)	84 (5)	120 (4)	105 (12)	101 (14)	113 (12)
	10-30	96 (11)	84 (7)	110 (87)	112 (6)	99 (2)	110 (8)
	30-60	91 (10)	117 (11)	130 (13)	129 (8)	99 (1)	138 (35)

Table A-23. Spoil Site. Trace element concentrations in parts per million dry weight (except Fe) in surface sediments (0-2 cm) from collections I (26 Dec. 1974 - 2 Jan. 1975), II (29 Jan. 1975) and III (19-20 Feb. 1975). Three replicates, one standard deviation ().

Element	Collections	Stations					
		1	2	3	4	5	6
Ag	I	1.2 (0.7)	0.4 (0.5)	1.8 (1.0)	0.9 (0.9)	0.5 (0.6)	0.3 (1.1)
	II	0.1 (0.5)	1.0 (1.5)	1.7 (0.6)	2.0 (1.5)	0.9 (1.0)	1.9 (1.1)
	III	0.8 (0.3)	0.4 (1.1)	0.4 (0.7)	1.4 (0.8)	0.9 (0.5)	0.1 (1.3)
As	I	6.3 (3.3)	8.0 (2.8)	7.7 (2.8)	8.3 (1.6)	5.1 (1.9)	7.7 (5.0)
	II	10.0 (2.0)	10.5 (1.1)	10.2 (1.5)	8.6 (1.1)	9.2 (2.3)	5.6 (2.6)
	III	3.1 (4.7)	6.7 (2.0)	7.9 (6.3)	9.8 (3.6)	9.0 (2.5)	9.9 (3.5)
Cd	I	2.4 (0.9)	-0.3 (1.1)	1.7 (0.1)	1.9 (1.2)	0.2 (0.7)	-0.9 (2.4)
	II	0.1 (0.8)	1.2 (1.2)	2.4 (1.2)	2.3 (1.4)	0.1 (0.4)	2.5 (0.1)
	III	1.0 (2.8)	0.3 (1.0)	0.3 (0.7)	-0.4 (1.5)	1.5 (1.6)	-1.2 (1.4)
Cr	I	163 (12)	208 (81)	195 (46)	199 (23)	187 (25)	241 (24)
	II	160 (29)	170 (16)	186 (30)	156 (24)	189 (5)	238 (5)
	III	217 (27)	214 (59)	204 (25)	222 (21)	186 (27)	230 (49)
Cu	I	32 (4)	50 (28)	50 (10)	41 (3)	41 (12)	51 (3)
	II	40 (17)	34 (3)	48 (8)	39 (3)	43 (1)	34 (7)
	III	42 (3)	46 (1)	52 (9)	54 (3)	42 (2)	46 (3)
Fe %	I	2.8 (.4)	2.6 (.5)	3.6 (.6)	3.4 (.1)	3.0 (.2)	4.0 (.2)
	II	3.0 (.7)	2.8 (.2)	3.7 (.4)	3.0 (.3)	3.3 (.03)	3.1 (.3)
	III	3.3 (.3)	3.5 (.1)	4.0 (.4)	4.0 (.4)	3.4 (.2)	3.8 (.3)

Table A-2 continued.

Element	Collections	Stations					
		1	2	3	4	5	6
Hg	I	.24 (.02)	.27 (.05)	.35 (.05)	.24 (.01)	.22 (.03)	.30 (.05)
	II	.22 (.03)	.23 (.02)	.31 (.03)	.27 (.01)	.29 (.01)	.28 (.02)
	III	.26 (.04)	.33 (.06)	.30 (.03)	.28 (.01)	.19 (.03)	.23 (.01)
Mn	I	382 (25)	333 (33)	460 (98)	410 (27)	399 (44)	533 (47)
	II	422 (121)	365 (25)	437 (37)	368 (26)	445 (20)	559 (130)
	III	447 (40)	560 (59)	568 (17)	518 (18)	492 (71)	541 (36)
Ni	I	89 (14)	96 (18)	119 (31)	107 (6)	88 (4)	126 (12)
	II	104 (26)	99 (5)	125 (16)	100 (14)	112 (5)	232 (78)
	III	96 (3)	176 (61)	159 (60)	115 (3)	99 (6)	110 (12)
Pb	I	29 (7)	29 (2)	39 (6)	31 (6)	34 (.4)	42 (9)
	II	33 (21)	22 (1)	36 (6)	32 (5)	30 (5)	35 (3)
	III	46 (19)	48 (7)	49 (4)	46 (8)	34 (9)	40 (6)
Se	I	1.0 (0.5)	0.8 (3.8)	1.8 (1.0)	2.4 (0.8)	1.8 (1.1)	3.4 (0.6)
	II	1.8 (1.0)	0.9 (1.0)	0.6 (1.3)	1.4 (0.7)	2.0 (1.7)	2.9 (1.3)
	III	1.1 (1.8)	3.8 (0.4)	4.0 (1.0)	2.2 (2.0)	3.5 (0.3)	2.4 (1.7)
Zn	I	88 (11)	97 (16)	120 (25)	105 (4)	99 (6)	132 (9)
	II	108 (36)	94 (10)	117 (12)	101 (13)	105 (3)	102 (12)
	III	109 (11)	124 (7)	139 (15)	139 (5)	113 (8)	129 (10)

Table A-24. Spoil Site. Trace element concentrations in parts per million dry weight (except Fe) in settled particulates in settling tubes from collections I (29 Jan. 1975), II (8 Feb. 1975) and III (17 Mar. 1975). Three replicates unless otherwise indicated, one standard deviation ().

Station:		1	2	3	4	5	6
Element	Collection						
Ag	I	1.0(1.2)	0.4(0.9)	1.7 ^{b)}	0.4 ^{b)}	1.1(0.5)	1.3(0.5)
	II	0.9(2.3) ^{a)}	1.4(1.0) ^{a)}	0.6(0.3)	1.3 ^{b)}	0.2(1.6)	c)
	III	-0.4(1.9) ^{a)}	-0.1(0.8) ^{a)}	0.7(1.8)	0.9(0.5)	0.9(0.6)	c)
As	I	10.6(3.8)	8.9(5.3)	13.8	8.8	13.4(1.0)	13.9(0.6)
	II	6.3(5.2)	7.2(2.2)	11.1(2.6)	10.4	15.3(4.0)	—
	III	10.1(4.9)	10.7(0.6)	7.8(0.2)	19.5(4.6)	11.2(4.8)	—
Cd	I	1.1(1.1)	0.8(2.6)	1.2	1.4	1.6(0.5)	2.2(1.0)
	II	0.9(0.2)	1.1(0.6)	-0.7(0.4)	1.4	0.1(1.6)	—
	III	0.3(1.2)	-0.9(1.1)	0.3(1.8)	1.7(1.6)	1.2(1.1)	—
Cr	I	245(32)	294(16)	184	198	224(9)	224(34)
	II	214(27)	222(71)	336(22)	221	259(12)	—
	III	205(3)	219(11)	266(28)	442(18)	245(30)	—
Cu	I	28(3)	62(8)	54	69	68(2)	70(11)
	II	46(2)	63(9)	66(3)	59	73(3)	—
	III	58(7)	58(11)	72(3)	130(14)	51(6)	—
Fe*	I	4.2(.5)	4.5(.3)	3.7	4.1	4.5(.2)	6.1(2.9)
	II	3.3(.1)	4.2(.6)	4.5(.1)	4.1	5.0(.2)	—
	III	4.2(.4)	4.0(.5)	4.7(.4)	8.9(.3)	4.1(.4)	—

TableA-24 continued

Station:		1	2	3	4	5	6
Element	Collection						
Hg	I	.43(.02)	.42(.01)	.62	.38	.53(.04)	.56(.07)
	II	.51(.03)	1.05(.06)	.54(.03)	.46	.42(.06)	—
	III	.32(.01)	.34(.02)	.39(.01)	.41(.04)	.31(.02)	—
Mn	I	917(105)	1191(312)	504	591	667(51)	696(118)
	II	960(154)	578(155)	1582(336)	576	770(53)	—
	III	600(85)	699(191)	685(49)	1174(38)	812(108)	—
Ni	I	350(94)	475(124)	117	132	157(4)	170(17)
	II	380(44)	136(13)	499(153)	132	161(2)	—
	III	140(6)	224(109)	145(13)	295(27)	340(91)	—
Pb	I	55(10)	58(6)	38	51	50(2)	54(9)
	II	49(12)	63(11)	59(8)	54	55(4)	—
	III	56(14)	45(1)	60(8)	105(9)	41(10)	—
Se	I	2.6(1.9)	2.8(1.1)	1.9	2.0	1.5(1.6)	2.1(0.9)
	II	3.5(0.1)	2.8(0.8)	2.7(1.1)	2.4	3.7(0.6)	—
	III	2.2(0.7)	2.9(0.2)	2.9(0.4)	4.0(2.3)	2.9(0.3)	—
Zn	I	142(14)	152(11)	137	138	159(4)	163(26)
	II	123(4)	163(26)	157(7)	152	172(9)	—
	III	143(12)	141(14)	159(16)	298(15)	138(10)	—

- a) Replicate sample number is two
b) Replicate sample number is one
c) Settling tubes at station 6
collections II and III were missing

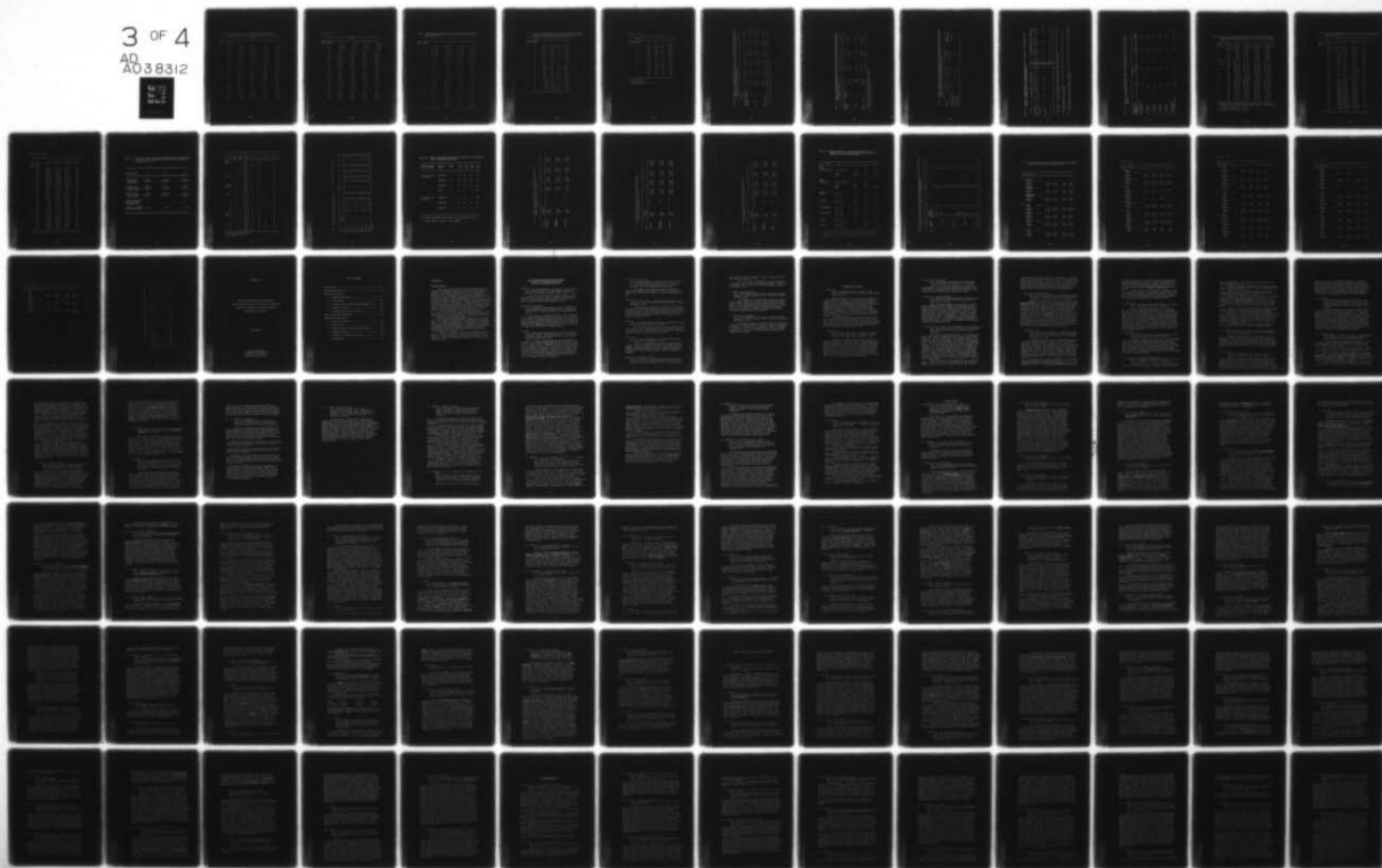
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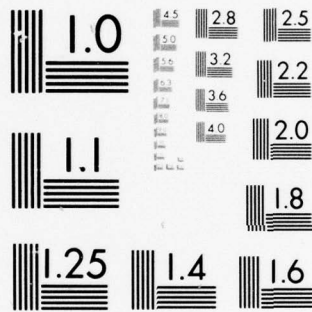


Table A-25. EBMUD. Trace element concentrations in parts per million dry weight (except Fe) in sediments at 4 depths from collection I (13 Dec. 1974). Three replicates, one standard deviation ().

Element	Depth (cm)	Station			
		1	2	3	4
Ag	0-2	0.6 (1.0)	1.1 (1.2)	1.6 (0.5)	1.3 (0.8)
	2-10	2.8 (.4)	17 (13)	2.1 (1.0)	2.2 (1.1)
	10-30	2.2 (.8)	2.2 (1.3)	1.2 (0.8)	3.0 (0.4)
	30-60	2.5 (.6)	2.6 (1.1)	2.1 (1.5)	5.6 (5.4)
As	0-2	14.5 (3.1)	18.0 (1.3)	12.8 (3.4)	10.7 (2.1)
	2-10	9.7 (4.7)	10.1 (2.8)	10.7 (3.6)	12.3 (1.5)
	10-30	6.2 (3.9)	9.8 (3.5)	10.3 (3.1)	9.6 (5.9)
	30-60	8.9 (5.4)	11.4 (1.1)	11.5 (4.3)	15.1 (6.0)
Cd	0-2	-.2 (.6)	1.5 (0.8)	1.1 (1.2)	0.9 (0.8)
	2-10	1.2 (.8)	0.1 (.2)	2.1 (0.7)	1.8 (0.4)
	10-30	2.7 (1.9)	0.3 (.7)	1.7 (1.2)	4.0 (1.0)
	30-60	2.3 (1.1)	2.4 (1.3)	2.0 (1.4)	1.7 (2.5)
Cr	0-2	268 (26)	253 (7)	217 (31)	291 (102)
	2-10	254 (59)	249 (35)	230 (18)	361 (32)
	10-30	317 (13)	285 (13)	229 (38)	364 (109)
	30-60	199 (74)	424 (78)	249 (57)	358 (25)
Cu	0-2	69 (.4)	78 (4)	68 (5)	66 (18)
	2-10	69 (9)	78 (2)	76 (6)	72 (12)
	10-30	56 (10)	72 (7)	68 (4)	59 (10)
	30-60	38 (15)	59 (20)	70 (2)	87 (24)
Fe %	0-2	4.6 (.3)	4.9 (.1)	4.2 (.2)	4.1 (.6)
	2-10	4.2 (.7)	4.6 (.3)	4.5 (.3)	4.5 (.3)
	10-30	3.8 (.4)	4.3 (.4)	4.0 (.3)	3.8 (.3)
	30-60	3.0 (1.4)	3.4 (.6)	4.1 (.5)	3.5 (.5)

Table A-25 continued

Element	Depth (cm)	1	Station 2	3	4
Hg	0-2	.49 (.01)	.55 (.01)	.61 (.07)	.55 (.06)
	2-10	.61 (.08)	.65 (.04)	.69 (.06)	.58 (.18)
	10-30	.58 (.06)	.67 (.03)	.86 (.16)	.62 (.19)
	30-60	.45 (.09)	.71 (.12)	.78 (.16)	1.40 (.10)
Mn	0-2	974 (74)	856 (24)	594 (59)	494 (89)
	2-10	531 (82)	497 (35)	476 (8)	578 (84)
	10-30	535 (56)	450 (27)	407 (33)	465 (31)
	30-60	378 (135)	423 (48)	441 (23)	436 (55)
Ni	0-2	267 (22)	148 (6)	135 (13)	135 (25)
	2-10	213 (58)	147 (6)	147 (10)	142 (14)
	10-30	176 (17)	131 (13)	131 (5)	127 (15)
	30-60	153 (59)	115 (20)	132 (17)	110 (22)
Pb	0-2	69 (8)	63 (4)	61 (5)	62 (20)
	2-10	63 (9)	59 (8)	64 (7)	59 (5)
	10-30	143 (45)	72 (4)	68 (17)	70 (15)
	30-60	39 (11)	89 (15)	61 (7)	122 (49)
Se	0-2	2.2 (0.7)	1.8 (1.7)	3.1 (0.9)	0.3 (1.0)
	2-10	1.5 (1.1)	1.4 (1.0)	1.2 (0.8)	1.5 (0.7)
	10-30	2.7 (1.7)	2.1 (1.6)	1.2 (0.5)	2.2 (1.4)
	30-60	1.8 (0.2)	1.2 (0.5)	0.1 (1.8)	2.5 (1.5)
Zn	0-2	179 (7)	186 (4)	166 (9)	159 (43)
	2-10	169 (20)	185 (12)	185 (14)	172 (12)
	10-30	154 (8)	192 (17)	181 (18)	196 (64)
	30-60	84 (27)	201 (41)	173 (23)	408 (330)

Table A-26. EBMUD. Trace element concentrations in parts per million dry weight (except Fe) in surface sediments (0-2 cm) from collections I (13 Dec. 1974) and II (17 Feb. 1975). Three replicates, one standard deviation ().

Element	Collection	Station			
		1	2	3	4
Ag	I	0.6 (0.9)	1.2 (1.3)	1.6 (0.5)	1.3 (0.8)
	II	0.5 (0.9)	-1.2 (0.3)	1.0 (0.8)	0.6 (1.5)
As	I	14.5 (3.1)	18.0 (1.3)	12.8 (3.4)	10.7 (2.0)
	II	9.5 (4.2)	14.3 (0.5)	12.8 (2.0)	8.2 (2.1)
Cd	I	-0.2 (0.6)	1.4 (0.8)	1.1 (1.2)	0.9 (0.8)
	II	-0.8 (1.4)	0.0 (2.3)	-0.2 (0.3)	1.3 (0.8)
Cr	I	268 (26)	253 (7)	217 (31)	291 (102)
	II	261 (25)	264 (32)	205 (26)	210 (36)
Cu	I	69 (.4)	78 (4)	68 (5)	66 (13)
	II	74 (3)	78 (1)	61 (6)	68 (5)
Fe %	I	4.6 (.3)	4.9 (.1)	4.2 (.2)	4.1 (.6)
	II	4.6 (.3)	4.8 (.2)	3.9 (.2)	4.0 (.1)
Hg	I	.49 (.01)	.55 (.01)	.61 (.07)	.55 (.06)
	II	.43 (.03)	.42 (.01)	.43 (.02)	.33 (.04)
Mn	I	974 (74)	856 (24)	594 (59)	494 (89)
	II	730 (43)	711 (132)	611 (64)	457 (10)
Ni	I	267 (22)	148 (6)	135 (13)	135 (25)
	II	276 (7)	168 (12)	130 (7)	129 (6)
Pb	I	69 (8)	63 (4)	61 (5)	62 (20)
	II	78 (19)	65 (11)	51 (7)	68 (8)
Se	I	2.2 (0.7)	1.8 (1.7)	3.1 (0.9)	0.3 (1.0)
	II	1.9 (1.0)	1.4 (0.6)	1.3 (0.7)	1.1 (1.2)
Zn	I	179 (7)	186 (4)	166 (9)	159 (43)
	II	171 (9)	182 (5)	144 (12)	154 (1)

Table A-27. EBMUD. Trace element concentrations in parts per million dry weight (except Fe) in settled particulates in settling tubes from collections I (10 Jan. 1975), II (11 Feb. 1975) and III (18 Mar. 1975). Three replicates unless otherwise indicated, one standard deviation ().

Station:		1	2	3	4
Element	Collection				
Ag	I	1.5(0.1) ^{a)}	1.2(1.3) ^{a)}	— ^{b)}	1.4(0.5)
	II	0.6(0.4)	0.5(0.4)	-0.7(0.8) ^{a)}	0.0(0.3)
	III	-0.4(0.3)	1.7(0.4) ^{a)}	— ^{b)}	0.9(0.7)
As	I	12.1(1.4)	12.7(0.4)	—	13.1(1.5)
	II	12.2(1.5)	9.4(4.5)	12.0(0.8)	8.3(7.8)
	III	13.5(1.2)	12.8(4.9)	—	17.7(1.8)
Ca	I	0.6(0.1)	0.5(0.4)	—	0.9(0.7)
	II	0.2(0.4)	0.2(0.7)	-0.7(0.5)	-0.1(1.3)
	III	-0.2(1.1)	1.1(0.2)	—	0.6(0.3)
Cr	I	231(20)	244(.1)	—	232(28)
	II	264(35)	253(7)	233(17)	247(73)
	III	219(29)	215(9)	—	274(41)
Cu	I	80(4)	93(6)	—	79(6)
	II	81(15)	75(19)	82(6)	71(29)
	III	80(18)	73(5)	—	94(9)
Fe%	I	4.7(.3)	5.3(.2)	—	4.9(.2)
	II	6.4(3.0)	4.6(.9)	5.0(.5)	4.6(1.7)
	III	4.6(.4)	4.5(.4)	—	7.2(2.9)
Hg	I	.76(.02)	.75(.05)	—	.93(.06)
	II	.66(.04)	.60(.04)	.60(.02)	.64(.02)
	III	.56(.06)	.47(.03)	—	.48(.01)

Table A-27 Continued

Station:		1	2	3	4
Element	Collection				
Mn	I	706(1)	752(37)	—	789(144)
	II	860(175)	950(124)	699(99)	788(248)
	III	561(54)	569(53)	—	780(94)
Ni	I	150(14)	171(19)	—	220(131)
	II	282(136)	347(141)	161(26)	265(83)
	III	142(21)	139(11)	—	199(29)
Pb	I	67(3)	80(5)	—	71(5)
	II	77(17)	69(8)	69(2)	72(22)
	III	62(5)	59(1)	—	72(11)
Se	I	2.4(1.8)	1.8(1.6)	—	2.0(1.3)
	II	3.3(0.6)	3.3(0.8)	0.8(0.3)	1.1(0.9)
	III	2.5(2.4)	1.0(1.0)	—	2.1(1.2)
Zn	I	178(7)	210(7)	—	191(12)
	II	199(37)	182(32)	198(21)	177(67)
	III	178(17)	175(14)	—	221(17)

a) Replicate number is two.

b) Settling tubes at Station 3, collections I and III were missing.

Table A-28. Spoil Site. Chlorinated hydrocarbon concentrations in parts per billion dry weight of surface sediments (0-2 cm) from collections I (20 Dec. 1974 - 2 Jan. 1975) and II (26 Jan. 1975). Three replicates, one standard deviation ().

Stations:		1	2	3	4	5	6
Compound	a) Collection						
p,p'-DDE	I	2.6 (3)	5.8 (3)	5.8 (1)	4.2 (2)	6.6 (8)	1.7 (.2)
	II	2.6 (3)	1.5 (.6)	2.0 (.1)	1.8 (.4)	1.6 (.2)	2.3 (.2)
p,p'-DDD	I	1.3 (.3)	1.5 (.2)	1.5 (.5)	1.4 (.1)	2.1 (.7)	1.3 (.1)
	II	5.4 (7)	2.7 (3)	1.8 (.6)	1.4 (.5)	1.6 (.1)	2.5 (.9)
PCB (Aroclor 1254)	I	23.2 (8)	23.4 (.7)	33.2 (2)	31.9 (3)	24.4 (4)	20.6 (1)
	II	13.4 (1)	13.9 (.5)	22.9 (3)	21.3 (3)	16.7 (2)	22.6 (1)

a) Traces of DDT were occasionally observed in these samples.

Table A-29. EBMUD. Chlorinated hydrocarbon concentrations in parts per billion dry weight of surface sediments (0-2 cm) from collections I (13 Dec. 1974) and II (17 Feb. 1975). Three replicates, one standard deviation ().

		1	2	3	4
Compound b)	Collection				
p,p'-DDE	I	11.0 (5)	17.3 (2)	5.1 (.3)	3.2 (1)
	II	10.2 (3)	12.6 (.7)	12.5 (.5)	7.3 (2)
p,p'-DDD	I	a)	—	—	—
	II	.5 (.03)	1.3 (.4)	1.2 (.1)	1.3 (.4)
PCB (Aroclor 1254)	I	83.3 (46)	126 (3)	31.0 (4)	24.1 (3)
	II	72.5 (7)	61 (15)	71.6 (8)	61.2 (6)

a) Trace only of p,p'-DDD in all Collection I samples.

b) No DDT was observed in these samples.

Table A-30. Oakland Inner Harbor and Dredge Hopper Sediments. Chlorinated hydrocarbon concentrations in parts per billion dry weight in sediments collected at the dredge site and from the dredge hoppers. Nine and 21 replicates, one standard deviation ().

Sample	% Moisture	n	p,p'-DDE	p,p'-DDD	PCB (Aroclor 1254)
Oakland Inner Harbor (30 Jan. 1975)	45	9	2.0 (.8)	2.3 (1)	58.9 (26)
Dredge Hopper (3-5 Feb. 1975)	56	21	18.4 (19)	4.7 (2)	130 (55)

Table A-31. Spoil Site. Suspended particulate loads in milligrams per liter, dry weight, of three particle size fractions. Replicate number n, one standard deviation ().

Collection/Date	n	Top Samples (3 m. below surface)				Bottom Samples (3 m. above bottom)			
		>4 μ	4 μ to 1 μ	1 μ to 0.1 μ	Total	>4 μ	4 μ to 1 μ	1 μ to 0.1 μ	Total
I ^{a)} 1 Jan. 1975 (pre-spoil)	4	4.7 (.4)	5.1 (.2)	2.2 (.2)	12.0 (.4)	5.9 (.3)	4.6 (.5)	2.5 (.2)	13.0 (.5)
II ^{b)} 3-5 Feb. 1975 (spoil)	4	3.9 (.2)	4.2 (.3)	1.7 (.2)	9.8 (.6)	7.5 (.8)	5.5 (.4)	2.3 (.3)	15.3 (1.3)
III ^{c)} 3 Mar. 1975	2	-	-	-	-	5.9 (1.1)	3.7 (.8)	2.0 (.1)	11.6 (2.0)

a) Samples were composites of equal volumes of water collected at outer and inner stations every six hours over a 24 hour period. Each of the samples (2 outer and 2 inner) consisted of 24 liters.

b) Samples were composites of equal volumes of water collected at outer and inner stations 1 hour & 4 hours after each disposal over a 42 hour period.

c) Samples were single 50 liter water composites collected at both inner stations and outer stations; equal volume composites were collected every 6 hours over a 24 hour period.

Table A-32. Spoil Site. Suspended particulate loads in milligrams per liter, dry weight, in 24 liter composite water samples collected 3-5 Feb. 1975. Two replicates in T+9, \pm one standard deviation.

Collection Time (Volume)	Particle Size				Total Load
	$>4 \mu$	4μ to 1μ	1μ to 0.1μ		
T-5 minutes before 1st disposal. (24 liters)	6.4	4.3	1.8		12.5
T + 0 hours (24 liters)	204.7	53.7	35.0		293.5
T + 1.5 hours (24 liters)	5.0	4.2	2.0		11.3
T + 3.0 hours (24 liters)	6.3	4.9	2.2		13.5
T + 4.5 hours (24 liters)	12.7	7.4	3.0		23.1
T + 9 hours after final disposal (2) 24 liter samples	13.3 ± 1.8	9.5 ± 0.3	5.5 ± 0.6		28.3 ± 0.9

Table A-33. Spoil Site. Trace element concentrations in parts per million dry weight (except Fe) in suspended particulates from collections I (8-9 Jan. 1975), II (3-5 Feb. 1975) and III (3-4 Mar. 1975). Four, 8, and 2 replicates for collections I, II and III, respectively, one standard deviation ().

Element	Collection	4u	4u to 1u	1u - 0.07u	Total
Ag	I	3.7 (1)	9.4 (1)	19.7 (20)	8.9 (5)
	II (spoil)	3.1 (2)	1.2 (1)	8.1 (3)	2.3 (1)
	III	0.8 (2)	6 (2)	11 (4)	4.1 (1)
As	I	9.5 (2)	10.5 (1)	11.5 (4)	10.2 (1)
	II	7.7 (1)	8.7 (2)	13.0 (2)	8.9 (1)
	III	9.0 (2)	8.2 (2)	7.4 (2)	8.3 (2)
Cd	I	9.5 (4)	3.5 (1)	21.2 (6)	8.9 (3)
	II	1.2 (1)	1.7 (1)	4.3 (2)	1.9 (1)
	III	1.2 (1)	3.4 (1)	5.0 (2)	2.6 (5)
Cu (Top) a)	I ^{b)}	136 (16)	244 (37)	304 (56)	222 (23)
	II ^{c)}	165 (13)	227 (19)	406 (28)	233 (7)
	III	-	-	-	-
Cu (Bottom) a)	I ^{b)}	133 (14)	177 (30)	272 (54)	160 (25)
	II ^{c)}	103 (11)	183 (11)	325 (12)	163 (12)
	III	104 (20)	211 (39)	293 (69)	170 (35)
Cr	I	211 (10)	255 (19)	317 (39)	246 (11)
	II	224 (8)	274 (14)	293 (16)	256 (8)
	III	204 (12)	224 (2)	214 (38)	212 (2)
Fe*	I	4.2 (.1)	5.5 (.1)	5.8 (.5)	5.0 (.1)
	II	4.3 (.1)	5.8 (.1)	5.9 (.2)	5.2 (.1)
	III	3.7 (.1)	4.9 (.1)	4.6 (.4)	4.2 (.1)
Hg	I ^{b)}	1.21 (.01)	1.43 (.34)	1.40	1.33
	II	0.97 (.10)	1.63 (.23)	2.43 (1.5)	1.39 (.15)
	III	0.85 (.01)	1.18 (.15)	1.52 (.07)	1.07 (.05)
Mn	I	924 (86)	1126 (12)	710 (83)	964 (51)
	II	981 (51)	1263 (65)	708 (19)	1047 (48)
	III	720 (15)	901 (45)	495 (7)	739 (30)
Ni	I	146 (1)	189 (3)	200 (20)	183 (9)
	II	167 (26)	275 (48)	219 (16)	216 (28)
	III	115 (2)	156 (6)	115 (5)	134 (4)
Pb	I	58 (1)	110 (10)	175 (15)	101 (7)
	II	77 (6)	117 (4)	245 (14)	118 (4)
	III	46 (1)	88 (2)	117 (9)	72 (2)
Se	I	3.5 (1)	3.1 (1)	4.4 (1)	3.4 (2)
	II	3.5 (1)	2.1 (1)	3.6 (1)	3.0 (1)
	III	2.2 (2)	3.4 (1)	4.3 (2)	3.3 (1)
Zn	I	155 (9)	215 (7)	205 (5)	189 (7)
	II	179 (12)	232 (5)	261 (16)	212 (8)
	III	147 (4)	185 (1)	179 (8)	164 (3)

a) Copper results from top and bottom waters listed separately since Cu concentrations were significantly ($P = 0.01$) different in top and bottom samples. This was not true of other metals.

b) The replicate sample number was 2 for collection I.

c) The replicate sample number was 4 for collection II.

Table A-34. Spoil Site. Trace element concentrations in parts per million dry weight (except Fe) from spoil site center suspended particulates collected 3-5 Feb. 1975. Time in hours after disposal, machine counting error ().

Size:		4u	4u to 1u	1u to 0.07u	Total
Element	Time				
Ag	T _{-0.08}	4.1 (1.5)	3.6 (1.5)	8.3 (3.2)	4.7
	T ₀ (in plume)	1.6 (1.4)	<1.4	<1.4	<1.4
	T _{1.5}	<1.4	2.3 (1.4)	4.5 (2.4)	2.3
	T ₃	<1.4	<1.4	6.1 (2.2)	1.7
	T _{4.5}	2.8 (1.4)	4.4 (1.5)	15.8 (2.3)	4.9
	T ₉	2.2 (1.5)	4.0 (1.5)	7.9 (1.5)	3.6
As	T _{-0.08}	12 (3)	6 (3)	17 (6)	10.6
	T ₀	10 (3)	6 (3)	7 (3)	8.9
	T _{1.5}	14 (3)	10 (3)	15 (6)	12.7
	T ₃	12 (3)	13 (3)	15 (5)	12.8
	T _{4.5}	12 (2)	9 (3)	18 (4)	11.8
	T ₉	14 (3)	7 (3)	7 (3)	10.7
Cd	T _{-0.08}	<1.5	7.2 (3.5)	14.8 (1.5)	9.1
	T ₀	2.0 (1.5)	<1.5	4.6 (1.6)	2.2
	T _{1.5}	<1.6	<1.5	<1.5	<1.5
	T ₃	<1.5	<1.5	2.6 (2.5)	<1.5
	T _{4.5}	1.7 (1.5)	3.1 (1.5)	5.6 (2.4)	2.6
	T ₉	<1.5	<1.5	6.8 (1.5)	<1.5
Cu	T _{-0.08}	134 (4)	270 (5)	464 (10)	223
	T ₀	113 (4)	171 (5)	230 (5)	136
	T _{1.5}	139 (4)	207 (5)	513 (9)	228
	T ₃	141 (4)	198 (4)	293 (6)	186
	T _{4.5}	90 (3)	153 (4)	335 (6)	140
	T ₉	103 (4)	180 (4)	293 (6)	157
Cr	T _{-0.08}	246 (18)	285 (22)	300 (23)	266
	T ₀	289 (18)	260 (21)	318 (22)	287
	T _{1.5}	245 (22)	245 (20)	335 (28)	260
	T ₃	226 (20)	222 (20)	235 (22)	226
	T _{4.5}	199 (18)	246 (19)	245 (22)	220
	T ₉	246 (21)	244 (21)	220 (23)	266
Fe	T _{-0.08}	3.69(0.01)	6.26(0.01)	5.37 (0.01)	4.8
	T ₀	3.81(0.01)	5.72(0.01)	6.51 (0.01)	4.4
	T _{1.5}	5.28(0.01)	5.59(0.01)	6.50 (0.01)	5.6
	T ₃	4.72(0.01)	5.40(0.01)	4.91 (0.01)	5.0
	T _{4.5}	3.88(0.01)	4.97(0.01)	6.02 (0.01)	4.5
	T ₉	4.80(0.01)	5.76(0.01)	6.37 (0.01)	5.3

Table A-34 Continued

Size:		4 μ	4 μ to 1 μ	1 μ to 0.07 μ	Total
Element	Time				
Mn	T _{-0.08}	1000 (21)	1271 (25)	628 (28)	1046
	T ₀	462 (17)	551 (20)	511 (21)	483
	T _{1.5}	1220 (25)	1191 (23)	775 (28)	1133
	T ₃	1100 (23)	1089 (22)	530 (22)	1007
	T _{4.5}	780 (19)	1023 (21)	690 (23)	849
	T ₉	1180 (23)	1320 (24)	778 (22)	1162
Hg ^{a)}	T _{-0.08}	0.68 (.04)	0.77 (.11)	0.88 (.08)	0.73
	T ₀	2.02 (.18)	4.16 (.03)	3.47 (.40)	3.04
	T _{1.5}	2.42 (.19)	2.38 (.16)	2.63 (.12)	2.44
	T ₃	7.88 (.19)	4.88 (.36)	2.53 (.58)	5.94
	T _{4.5}	1.46 (.10)	15.32 (2.3)	6.60 (.50)	6.51
	T ₉	0.61 (.03)	1.38 (.06)	1.91 (.03)	1.16
Ni	T _{-0.08}	155 (8)	217 (6)	323 (7)	265
	T ₀	98 (5)	184 (6)	198 (6)	124
	T _{1.5}	178 (6)	186 (6)	217 (8)	187
	T ₃	148 (6)	175 (6)	163 (7)	160
	T _{4.5}	104 (5)	149 (5)	198 (7)	130
	T ₉	146 (5)	186 (6)	205 (6)	168
Pb	T _{-0.08}	63 (8)	134 (3)	243 (18)	110
	T ₀	61 (8)	138 (8)	211 (9)	91
	T _{1.5}	67 (9)	105 (8)	317 (15)	123
	T ₃	69 (9)	107 (8)	219 (13)	107
	T _{4.5}	48 (8)	92 (8)	179 (11)	78
	T ₉	55 (8)	101 (8)	181 (8)	89
Se	T _{-0.08}	4.8 (1.8)	5.4 (2.0)	8.2 (4.0)	5.4
	T ₀	3.4 (1.7)	3.4 (1.8)	3.4 (1.8)	3.4
	T _{1.5}	3.1 (2.1)	<1.8	4.1 (3.1)	2.7
	T ₃	4.5 (2.0)	2.4 (1.8)	4.9 (2.7)	3.8
	T _{4.5}	3.5 (1.8)	2.7 (1.7)	3.6 (2.4)	3.2
	T ₉	3.4 (1.8)	1.9 (1.8)	3.4 (1.8)	2.9
Zn	T _{-0.08}	182 (4)	251 (4)	220 (5)	210
	T ₀	153 (4)	263 (5)	290 (5)	188
	T _{1.5}	215 (4)	224 (4)	310 (6)	234
	T ₃	185 (4)	205 (4)	189 (5)	192
	T _{4.5}	151 (4)	194 (4)	241 (5)	176
	T ₉	185 (4)	212 (4)	252 (5)	213

Table A-35. Spoil Site. Soluble trace metal concentrations in milligrams per liter at Spoil Site Center and stations. Replicate numbers, 95% confidence limits ().

Collection/Date	Pb	Cd	Cu
Collection II plume samples 4 Feb. 1975	1.88 (0.98-2.78) 4	1.35 (1.26-1.44) 4	5.0 (3.64-6.42) 4
Collection II station samples 3-4 Feb. 1975	0.28 (0.02-0.54) 4	0.22 (0.02-0.42) 4	1.17 (0.53-1.81) 4
Collection III station samples 3 Mar. 1975	0.14 (0.07-0.27) 3	0.23 (0.09-0.38) 3	1.42 (0.91-1.93) 4
Factor increase in plume above ambient or natural conc.	9	6	4
Duration of increase at Spoil Site Center	<< 3 hr.	<< 3 hr.	<< 3 hr.

Table A-36. Spoil Site. Chlorinated hydrocarbon concentrations in nanograms per liter in water samples.

Date	Station no.	Sample no.	Depth meters	Sampling time	P,p'-DDE	P,p'-DDD	P,p'-DDT ^{a)}	PCB Aroclor 1254
Prespoil 8 Jan. 1975	5	1	6	1535-1605	.06	.06		.70
	5	2	2	1630-1705	.05	.04		.52
	2	3	6	1807-1840	.19	.10		2.0
	3	4	5.5	1915-2015	.04	.05		.40
	6	5	7	2053-2200	.17	.08		1.1
	1	6	6	2255-0000	.08	.06		.88
9 Jan. 1975	4	7	6	0030-0140	.12	.08		.85
	5	8	6	0215-0320	.14	.12		1.3
	2	9	7	0355-0455	.12	.06		.99
	3	10	6	0530-0635	.07	.08		.76
	6	11	6	0715-0815	.13	.07		1.0
	6	12	6	0815-0915	.13	.07		1.0
Spoiling Period 3 Feb. 1975	6	1	5	1750-1830	.17	.28		1.7
	6	2 ^{b)}	5	1840-2010	.12	.26		1.0
	6	3	5	2230-2310	.05	.11		.41
	6	4 ^{b)}	5	0045-2100	.12	.29	.17	1.0
	6	5	5	0235-0400	.07	.13		.66
	4	6 ^{b)}	5	0710-0750	.06	.12	.10	.81
4 Feb. 1975	2	7	5	1300-1335	.08	.21	.15	.74
	SS ^{c)}	8 ^{b)}	5	1350-1425	.44	.98		6.6
	SS	9	5	1505-1540	.06	.15	.09	.45
	SS	10	5	1755-1825	.10	.28	.19	.76
	SS	11 ^{b)}	5	1940-2020	.14	.39		2.1
	SS	12	5	2030-2137	.08	.18		.61
Post Spoil 3 Mar. 1975	SS	13	5	2145-2222	.09	.22		.91
	SS	1	3	1415-1515	.16	.40		1.0
	SS	2	8	1520-1620	.25	.58		2.3
	SS	3	3	1915-2015	.05	.19	.43	.61
	SS	4	8	2025-2120	.14	.44	.66	1.6
	SS	5	3	0220-0315	.06	.28	.71	.85
4 Mar. 1975	SS	6	8	0325-0430	.15	.18	.20	1.2
	SS	7	3	0715-0815	.04	.11	.24	.58
	SS	8	8	0820-0915	.04	.14		.51

a) empty spaces indicate compound not detected

b) sample taken during disposal operation

c) Spoil Site Center

Table A-37. EBMUD. Trace element concentrations in parts per million dry weight (except Fe) and in parts per billion wet weight () in suspended particulates from EBMUD sewage treatment plant.

Sample/Date	Particulate Load mg/l	Elements											
		Ag	As	Cd	Cr	Cu	Fe%	Hg	Mn	Ni	Pb	Se	Zn
ERMUD - 1 20-27 Nov. 1974	126.4	38 (4.8)	<1	86 (10.8)	661 (83.6)	2793 (353)	3.6	5.34 (.68)	662 (83.7)	141 (17.8)	1367 (172)	<1	4760 (602)
ERMUD - 2 17-22 Jan. 1975	67.8	7 (.5)	6 (.5)	6 (.5)	1616 (109)	6521 (442)	2.5	- a)	336 (22.8)	176 (11.9)	2047 (139)	4.8 (.3)	10709 (726)
ERMUD - 3 29 Jan. - 2 Feb. 1975	78.1	<1	7 (.5)	<1 (.5)	1219 (95.1)	5140 (400)	2.4	7.30 (.57)	355 (27.7)	178 (13.9)	2178 (170)	<1	9945 (775)
ERMUD - 4 5-12 Feb. 1975	70.0	1.0 (.1)	9 (.5)	1 (.1)	258 (18.1)	112 (7.8)	4.0	8.44 (.59)	1010 (70.7)	363 (25.4)	60 (4.2)	2.2 (.2)	170 (11.9)
ERMUD - 5 13-19 Feb. 1975	76.2	<1	6 (.5)	<1 (.5)	877 (66.6)	4128 (313)	2.1	6.38 (.49)	282 (21.5)	126 (9.6)	1419 (108)	<1	3233 (246)
ERMUD - 6 20-26 Feb. 1975	78.8	71 (5.6)	8 (.5)	86 (6.7)	1882 (148)	4293 (338)	3.7	7.54 (.59)	506 (39.9)	135 (10.6)	1670 (131)	3.7 (.3)	7195 (566)
ERMUD - 7 27 Feb - 5 Mar. 1975	67.5	7 (.5)	15 (1.0)	8 (.6)	1272 (86)	4741 (320)	2.9	11.37 (.77)	510 (34.4)	174 (11.7)	1273 (86)	8.1 (.5)	6476 (437)
ERMUD - 8 7-13 Mar. 1975	68.5	63 (4.3)	8 (.5)	67 (4.8)	1036 (70)	3339 (227)	2.2	12.19 (.84)	307 (21.0)	120 (8.2)	1152 (78)	7.0 (.4)	3411 (232)

a) sample lost

Table A-38. EBMUD. Chlorinated hydrocarbon concentrations-in nanograms per liter from sewage water samples.

Sampling Dates, Ave. flow (mgd)	Type of sample	Sample No.	p,p'- DDE	p,p'- DDD	p,p'- DDT	PCB 1254
5-13 Dec. 1974 87.9 ± 18.4	Composite	1	40	60	90	550
	Composite	2	30	60	90	730
	Composite	3	30	60	80	430
	Grab ^{a)}	4	b)	b)	b)	1020
10-20 Feb. 1975 101 ± 29	Composite	1	10	b)	10	410
	Composite	2	10	b)	10	340
	Composite	3	10	b)	10	270

a) Sample collected by method described in Schmidt et al. (1971).

b) No DDD (<10ppt) observed in these samples.

Table A-39. Laboratory Uptake Study. Cadmium concentrations in parts per million dry weight. Three replicates, one standard deviation (), 8, 10, and 12 individuals per replicate of M. nasuta, P. californiensis and M. edulis, respectively.

Species	Assumed Concentration	Days					Exposure		
		0	1	3	7	15			
<u>Macoma nasuta</u>	control	1.4 (.5)	1.8 (.2)	1.7 (.5)	1.0 (.1)	1.4 (.1)			
	low		.4 (.4)	1.0 (1.1)	.9 (.2)	1.0 (.6)			
	medium		1.4 (1.1)	1.5 (1.0)	1.7 (.2)	2.2 (.2)			
	high		2.1 (.5)	1.5 (.1)	2.1 (.2)	2.7 (.2)			
<u>Pectinaria californiensis</u>	control	3.7 (.9)	3.2 (.8)	3.3 (.5)	2.7 (.2)	2.0 (.7)			
	low		3.1 (.3)	3.6 (.8)	2.6 (.7)	1.8 (1.8)			
	medium		3.7 (.9)	2.6 (.9)	1.0 (.6)	2.3 (1.2)			
	high		3.5 (1.0)	2.9 (.5)	2.8 (.4)	2.7 (.2)			
<u>Mytilus edulis</u>	control	5.7 (1.1)	5.8 (.4)	6.2 (.7)	5.2 (1.1)	6.9 (.3)			
	low		5.4 (1.6)	6.1 (1.1)	6.8 (1.1)	7.0 (1.1)			
	medium		6.0 (.9)	6.3 (.6)	7.4 (.5)	9.5 (.6)			
	high		7.2 (.2)	8.3 (1.0)	13.8 (2.2)	21.0 (2.7)			

Table A-40. Laboratory Uptake Study. Mercury concentrations in parts per million dry weight. Three replicates, one standard deviation (), 8, 10, and 12 individuals per replicate for M. nasuta, P. californiensis, and M. edulis, respectively.

Species	Assumed Concentration	Days Exposure				
		0	1	3	7	15
<u>Macoma nasuta</u>	control					
	low	.40 (.03)	.44 (.11)	.46 (.11)	.41 (.04)	.37 (.03)
	medium		.54 (.02)	.52 (.04)	.55 (.03)	.51 (.02)
	high		.63 (.09)	.67 (.05)	1.00 (.09)	1.49 (.06)
<u>Pectinaria californiensis</u>	control					
	low	.08 (.02)	.09 (.01)	.09 (.04)	.06 (.01)	.05 (.01)
	medium		.14 (.04)	.18 (.04)	.16 (.03)	.20 (.05)
	high		.13 (.05)	.24 (.02)	.40 (.07)	.47 (.06)
<u>Mytilus edulis</u>	control					
	low	.48 (.03)	.44 (.03)	.50 (.03)	.44 (.06)	.37 (.14)
	medium		.56 (.04)	.79 (.08)	1.17 (.15)	1.90 (.05)
	high		.77 (.06)	1.68 (.10)	3.34 (.31)	4.83 (1.89)
			1.43 (.17)	3.33 (.40)	5.27 (.52)	8.52 (.97)

Table A-41. Laboratory Uptake Study. Lead concentrations in parts per million dry weight. Three replicates, 1 standard deviation (), 8, 10, and 12 individuals per replicate for M. nasuta, P. californiensis, and M. edulis, respectively.

Species	Assumed Concentration	Days Exposure				
		0	1	3	7	15
<u>Macoma nasuta</u>	control	6.1 (1.6)	5.5 (.3)	8.7 (.6)	3.6 (1.1)	5.9 (2.0)
	low		6.8 (1.1)	9.3 (.3)	5.0 (.8)	5.2 (.1)
	medium		9.3 (1.0)	7.5 (1.2)	5.4 (1.0)	4.6 (1.4)
	high		6.0 (2.0)	7.2 (1.3)	8.2 (.5)	12.5 (1.0)
<u>Pectinaria californiensis</u>	control	3.6 (2.2)	5.6 (1.8)	3.7 (2.0)	3.1 (1.5)	1.7 (1.7)
	low		3.6 (.4)	3.4 (.8)	4.9 (1.2)	2.0 (1.7)
	medium		3.3 (1.3)	4.0 (.3)	6.4 (1.6)	3.6 (1.4)
	high		3.7 (.3)	4.3 (2.2)	4.8 (1.9)	4.7 (1.5)
<u>Mytilus edulis</u>	control	5.0 (1.8)	4.5 (.5)	5.1 (1.1)	4.8 (.5)	3.8 (.4)
	low		5.2 (.9)	6.6 (1.0)	6.2 (1.1)	6.7 (.8)
	medium		4.9 (1.1)	5.3 (.8)	9.9 (1.9)	9.8 (1.0)
	high		7.2 (.3)	10.4 (1.1)	15.5 (1.9)	22.3 (2.9)

Table A-42. Laboratory Uptake Study. Soluble trace metal concentrations in micrograms per liter in composited water samples from the inflow side and from the outflow side of the same tanks.

Group	Element	Tank	Cd	Hg	Pb
<u>Inflow</u>					
(Assumed Concentrations) ^{a)}					
		Control (seawater)	0.03	0.1	0.2
		Low	0.09	0.3	0.6
		Medium	0.27	0.9	1.8
		High	4.0	2.7	5.4
<u>Inflow</u>					
(Observed Concentrations)					
		Control (seawater)	-	0.01	-
		Low	≤0.1	.22	1.7
		Medium	0.4	1.2	3.6
		High	4.5	3.9	8.6
<u>Outflow</u>					
<u>M. nasuta</u>		Day 1 Low	.3	.11	.84
		Day 7 Low	.3	.05	.54
<u>M. nasuta</u>		Day 1 High	6.3	1.1	3.6
		Day 7 High	4.5	.9	2.8
<u>P. californiensis</u>		Day 1 Low	.3	.11	.96
		Day 7 Low	≤.1	.12	.2
<u>P. californiensis</u>		Day 1 High	<.1	3.9	<.1
		Day 7 High	<.1	1.6	.1
<u>M. edulis</u>		Day 1 Low	<.1	.12	.9
		Day 7 Low	≤.1	.16	.2
<u>M. edulis</u>		Day 1 High	.1	2.9	<.1
		Day 7 High	.1	1.7	<.1

a) The assumed concentrations of Cd, Hg and Pb in the seawater control were based on literature values (see discussion). Higher concentrations, created by adding measured amounts of metal salts, were calculated using these assumed values for seawater.

Table A-43. Laboratory Uptake Study. Trace metal concentrations in parts per million dry weight in silica sand, and sediments from cores in surface samples from P. californiensis tanks.

Element:				Cd	Hg	Pb
Tank	Days Exposure	Assumed Exposure Concentration				
1	0	Control	3.4		0.00	0.0
	7	Control	2.7		0.01	7.7
	15	Control	0.5		0.01	2.2
	15(surface)	Control	1.9		0.01	3.1
2	0	Low	2.6		0.02	0.0
	7	Low	3.1		0.02	5.9
	15	Low	1.3		0.01	0.0
	15(surface)	Low	2.9		0.01	6.7
3	0	Medium	3.9		0.01	0.0
	7	Medium	3.1		0.01	0.0
	15	Medium	1.2		0.01	2.7
	15(surface)	Medium	2.7		0.01	0.0
4	0	High	3.1		0.03	3.7
	7	High	2.2		0.02	4.3
	15	High	3.3		0.01	5.4
	15(surface)	High	4.1		0.04	7.5

Table A-44. Field-Laboratory Study. Trace element concentrations in parts per million in *M. edulis* local to Oakland Inner Harbor, and *M. edulis* and *C. gigas* transplanted from Tomales Bay. Five replicates, one standard deviation (), 5 individuals per replicate.

Element	Species/Chamber	Days Exposure				
		0	1	3	7	15
<hr/>						
<u>Ag</u>						
	<u>Mytilus edulis:</u> (local)	0.9 (.2)				
	Outside		1.0 (.6)	.6 (.2)	1.0 (.4)	.8 (.3)
	Unfiltered		.9 (.2)	1.2 (.4)	1.0 (.4)	.9 (.4)
	Filtered		.9 (.5)	1.3 (.5)	.5 (.3)	.9 (.6)
	<u>Mytilus edulis:</u> (transplants)	1.3 (.5)				
	Outside		.8 (.3)	1.1 (.3)	.9 (.3)	.7 (.5)
	Unfiltered		1.3 (.7)	1.6 (.4)	1.2 (.6)	1.6 (.4)
	Filtered		.9 (.5)	.7 (.7)	.8 (.2)	.5 (.6)
	<u>Crassostrea gigas:</u> (transplants)	2.1 (.6)				
	Outside		1.6 (1.0)	2.5 (.5)	2.1 (.5)	4.2 (.5)
	Unfiltered		2.2 (.4)	1.5 (.2)	1.9 (.4)	3.1 (.4)
	Filtered		2.4 (1.0)	2.1 (.7)	1.5 (.7)	2.2 (.5)
<u>As</u>						
	<u>M. edulis</u> (local)	5.6 (1.5)				
	Outside		7.7 (2.2)	6.7 (1.4)	5.2 (.9)	5.7 (3.5)
	Unfiltered		6.9 (2.1)	7.0 (.8)	5.0 (1.4)	6.3 (1.6)
	Filtered		5.4 (.7)	6.4 (1.3)	5.2 (.5)	5.4 (1.7)
	<u>M. edulis</u> (transplants)	5.5 (.9)				
	Outside		4.2 (.6)	4.3 (.4)	4.3 (.3)	4.5 (.6)
	Unfiltered		4.1 (.5)	4.2 (.8)	3.9 (.6)	4.6 (.2)
	Filtered		4.4 (.8)	3.8 (.6)	3.9 (.7)	4.4 (.4)
	<u>C. gigas</u> (transplants)	5.5 (.5)				
	Outside		5.8 (.4)	6.2 (.7)	5.4 (.6)	6.6 (.7)
	Unfiltered		6.4 (.7)	5.4 (.7)	5.6 (.8)	6.3 (.6)
	Filtered		5.9 (.4)	5.8 (.6)	5.8 (.6)	6.3 (.8)
<u>Cd</u>						
	<u>M. edulis</u> (local)	5.5 (1.1)				
	Outside		7.0 (1.2)	5.4 (1.0)	5.8 (.9)	7.0 (.6)
	Unfiltered		6.9 (1.8)	6.4 (.7)	6.1 (.8)	5.9 (1.0)
	Filtered		6.8 (1.4)	7.3 (1.8)	7.4 (1.4)	6.6 (2.4)

Table A-44 continued

Element	Species/Chamber	0	Days		Exposure	
			1	3	7	15
	<u>M. edulis</u> (transplant)	6.4 (1.1)				
	Outside		4.5 (.9)	4.5 (.5)	4.3 (.6)	3.1 (.9)
	Unfiltered		4.3 (1.2)	4.4 (1.2)	4.4 (.7)	4.0 (.5)
	Filtered		4.8 (.7)	4.7 (.8)	4.0 (1.3)	3.3 (.4)
	<u>C. gigas</u> (transplant)	12.1 (2.1)				
	Outside		9.7 (1.7)	10.6 (1.5)	10.0 (1.9)	13.1 (1.7)
	Unfiltered		10.7 (1.4)	9.1 (1.4)	10.6 (1.3)	13.0 (1.8)
	Filtered		11.2 (1.2)	11.6 (1.4)	9.5 (.9)	12.6 (.7)
Cr	<u>M. edulis</u> (local)	<1				
	Outside		<1	<1	2.1 (1.7)	1.4 (1.2)
	Unfiltered		<1	2.1 (1.4)	<1	<1
	Filtered		1.5 (2.2)	<1	<1	<1
	<u>M. edulis</u> (transplant)	<1				
	Outside		<1	<1	1.7 (1.7)	<1
	Unfiltered		<1	<1	1.3 (1.4)	<1
	Filtered		<1	<1	<1	<1
	<u>C. gigas</u> (transplant)	2.4 (1.6)				
	Outside		2.5 (2.9)	2.2 (2.7)	2.7 (1.4)	4.3 (1.7)
	Unfiltered		2.3 (2.0)	3.2 (3.1)	2.4 (1.7)	3.5 (1.2)
	Filtered		4.0 (1.2)	3.5 (1.7)	2.5 (1.9)	3.6 (2.0)
Cu	<u>M. edulis</u> (local)	6.3 (1.1)				
	Outside		8.1 (.8)	5.6 (1.2)	6.1 (.9)	5.6 (3.1)
	Unfiltered		6.9 (1.5)	8.0 (.5)	6.9 (1.3)	8.5 (1.6)
	Filtered		5.4 (.2)	5.8 (.9)	5.7 (.4)	5.4 (.9)
	<u>M. edulis</u> (transplant)	7.5 (1.1)				
	Outside		7.0 (.5)	7.1 (1.5)	6.3 (1.0)	7.5 (.7)
	Unfiltered		8.2 (1.0)	8.7 (1.3)	8.6 (1.4)	16.6 (1.7)
	Filtered		7.1 (1.1)	5.7 (.6)	5.7 (.6)	9.4 (1.4)
	<u>C. gigas</u> (transplant)	81 (12)				
	Outside		58 (12)	74 (14)	80 (19)	111 (7)
	Unfiltered		56 (5)	52 (6)	66 (9)	93 (11)
	Filtered		69 (18)	62 (8)	71 (10)	106 (13)

Table A-44 continued

Element	Species/Chamber	Days Exposure				
		0	1	3	7	15
<u>Fe</u>	<u>M. edulis</u> (local)	78 (13)				
	Outside		100 (26)	80 (15)	87 (9)	59 (33)
	Unfiltered		87 (19)	77 (5)	73 (7)	73 (10)
	Filtered		81 (12)	74 (14)	83 (8)	62 (17)
	<u>M. edulis</u> (transplant)	73 (5)				
	Outside		90 (10)	80 (18)	86 (7)	112 (46)
	Unfiltered		102 (6)	83 (14)	84 (12)	93 (13)
	Filtered		99 (11)	76 (4)	83 (21)	97 (41)
	<u>C. gigas</u> (transplant)	180 (16)				
	Outside		194 (28)	182 (32)	170 (31)	212 (15)
	Unfiltered		182 (29)	154 (27)	186 (13)	200 (14)
	Filtered		219 (45)	180 (23)	187 (21)	243 (37)
<u>Hg</u>	<u>M. edulis</u> (local)	.29 (.08)				
	Outside		.39 (.03)	.28 (.05)	.30 (.03)	.30 (.04)
	Unfiltered		.32 (.06)	.28 (.01)	.30 (.02)	.32 (.05)
	Filtered		.29 (.06)	.34 (.07)	.37 (.02)	.35 (.16)
	<u>M. edulis</u> (transplant)	.26 (.01)				
	Outside		.24 (.02)	.26 (.02)	.29 (.01)	.29 (.03)
	Unfiltered		.28 (.02)	.25 (.02)	.24 (.02)	.25 (.02)
	Filtered		.27 (.03)	.26 (.03)	.25 (.02)	.30 (.04)
	<u>C. gigas</u> (transplants)	.25 (.04)				
	Outside		.19 (.02)	.21 (.06)	.26 (.05)	.30 (.04)
	Unfiltered		.21 (.01)	.22 (.01)	.22 (.04)	.21 (.02)
	Filtered		.22 (.03)	.24 (.03)	.21 (.02)	.30 (.05)
	<u>Mn</u>					
	<u>M. edulis</u> (local)	5.4 (1.2)				
	Outside		6.7 (2.3)	7.8 (3.7)	5.4 (2.2)	4.8 (3.3)
	Unfiltered		7.3 (4.4)	7.4 (1.6)	6.6 (1.7)	4.1 (1.4)
	Filtered		5.8 (2.8)	4.0 (.9)	8.3 (4.0)	3.5 (2.1)
	<u>M. edulis</u> (transplants)	6.3 (1.7)				
	Outside		7.7 (3.5)	7.3 (2.2)	8.8 (2.5)	6.9 (.8)
	Unfiltered		10.0 (1.7)	8.1 (4.5)	7.6 (2.2)	7.5 (1.7)
	Filtered		10.0 (2.6)	7.1 (1.9)	4.8 (1.1)	6.4 (2.9)
	<u>C. gigas</u> (transplants)	54 (4)				
	Outside		48 (5)	45 (7)	45 (13)	55 (12)
	Unfiltered		51 (7)	40 (9)	55 (15)	49 (6)
	Filtered		58 (15)	58 (13)	43 (13)	62 (9)

Table A-44 continued

Element	Species/Chamber	Days Exposure				
		0	1	3	7	15
<u>Ni</u>	<u>M. edulis</u> (local)	<1				
	Outside		1.2 (.2)	1.2 (1.2)	<1	<1
	Unfiltered		<1	1.3 (.6)	<1	<1
	Filtered		<1	<1	<1	<1
	<u>M. edulis</u> (transplant)	1.2 (.5)				
	Outside		1.0 (.4)	1.0 (.9)	1.4 (.4)	<1
	Unfiltered		1.0 (.3)	1.3 (.7)	1.2 (.7)	1.6 (.9)
	Filtered		1.5 (.5)	<1	1.2 (.7)	1.6 (.4)
	<u>C. gigas</u> (transplant)	1.2 (.5)				
	Outside		1.1 (.7)	<1	<1	1.6 (.5)
	Unfiltered		<1	<1	<1	1.3 (.8)
	Filtered		<1	<1	<1	<1
	<u>Pb</u>					
	<u>M. edulis</u> (local)	2.8 (.5)				
	Outside		2.4 (.6)	2.1 (.6)	3.2 (1.3)	1.6 (.9)
	Unfiltered		3.0 (1.3)	2.9 (.8)	2.9 (.6)	3.0 (1.2)
	Filtered		3.1 (1.0)	2.9 (1.4)	3.8 (1.5)	2.6 (1.2)
	<u>M. edulis</u> (transplant)	.9 (.3)				
	Outside		2.9 (1.1)	2.8 (1.3)	2.3 (.7)	2.5 (.4)
	Unfiltered		2.5 (.5)	2.9 (.6)	2.2 (.7)	2.1 (1.0)
	Filtered		3.3 (1.2)	3.6 (2.9)	2.8 (.5)	2.5 (.5)
	<u>C. gigas</u> (transplant)	1.2 (.8)				
	Outside		1.5 (.2)	1.2 (1.0)	1.3 (.6)	2.3 (1.0)
	Unfiltered		.8 (.6)	1.7 (.6)	2.6 (1.0)	1.7 (.7)
	Filtered		1.6 (.8)	1.5 (1.1)	1.2 (.9)	1.7 (.3)
<u>Se</u>	<u>M. edulis</u> (local)	4.2 (.9)				
	Outside		5.0 (1.0)	4.3 (.6)	3.8 (.4)	3.6 (2.1)
	Unfiltered		5.3 (1.4)	4.4 (.4)	4.3 (.4)	4.5 (.7)
	Filtered		4.0 (.8)	4.6 (.9)	4.9 (1.3)	3.8 (1.1)
	<u>M. edulis</u> (transplant)	5.4 (.5)				
	Outside		4.8 (.5)	4.8 (.7)	4.4 (.7)	4.8 (.5)
	Unfiltered		5.0 (.8)	4.6 (.4)	4.1 (.5)	5.2 (.8)
	Filtered		5.1 (.7)	4.7 (.3)	4.5 (.9)	4.3 (.3)
	<u>C. gigas</u> (transplant)	3.1 (.3)				
	Outside		3.3 (.3)	3.5 (.5)	3.7 (.4)	3.8 (.3)
	Unfiltered		3.6 (.5)	3.1 (.4)	3.3 (.3)	3.7 (.5)
	Filtered		3.4 (.2)	3.2 (.3)	3.0 (.2)	3.4 (.5)

Table A-44 continued

Element	Species/Chamber	0	1	Days		Exposure		15
				3	7	7	15	
<u>Zn</u>	<u>M. edulis</u> (local)	131 (31)						
	Outside		147 (45)	124 (21)	123 (10)		124 (77)	
	Unfiltered		154 (23)	166 (43)	134 (34)		128 (18)	
	Filtered		146 (18)	137 (38)	146 (22)		124 (54)	
	<u>M. edulis</u> (transplant)	82 (11)						
	Outside		121 (34)	127 (31)	147 (29)		159 (14)	
	Unfiltered		149 (23)	130 (40)	132 (18)		132 (30)	
	Filtered		120 (17)	119 (22)	121 (10)		172 (17)	
	<u>C. gigas</u> (transplant)	397 (38)						
	Outside		367 (54)	441 (88)	470 (86)		656 (31)	
	Unfiltered		361 (57)	334 (38)	436 (32)		586 (46)	
	Filtered		402 (68)	394 (29)	426 (47)		659 (70)	

Table A-45. Field-Laboratory Study. Trace element concentrations in parts per million dry weight (except Fe) in suspended particulates collected 12 Dec. 1974.

Water Source	Particulate Load mg/l a)	Elements							
		As	Cr	Cu	Fe (%)	Mn	Ni	Pb	Zn
Outside	13.5	10	210	80	4.2	1060	153	89	260
Unfiltered	9.7	7	173	70	4.2	910	145	87	203
Filtered	5.0	16	263	76	5.0	1070	168	74	230

a) Estimated standard deviation is 1.5 mg/l of suspended particulates.

APPENDIX I-B

Annotated and Supplementary Bibliographies
of the Literature Pertinent to the Effects of Dredging and
Dredge Spoil Disposal on Pollutant Availability
to Estuarine Invertebrates

prepared by

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Introduction

Literature search

Since relatively few papers have considered the effects of dredging and dredge spoil disposal on pollutant availability to estuarine invertebrates, papers relevant to one or more aspects of dredging and/or pollutant availability have been selected for inclusion in this bibliography. Many of the references were obtained by computer searches of Water Resources Abstracts at Lawrence Berkeley Laboratory, through RECON Information Service, and searches of Biological Abstracts and Science Citation Index through Bioscience Information Service in Philadelphia. A computer search of current entries into Biological and Chemical Abstracts was conducted through the Center for Information Services, University of California, Los Angeles. Manual searching was also conducted by use of various abstracts, other bibliographies, reviews, and references.

Of the more than two thousand references gathered, 175 were chosen for annotation because they were particularly relevant to an understanding of pollutant availability. Another 406 are included in a supplementary bibliographic section.

The annotated and supplementary bibliographies are divided into general subject categories which are arranged in the following order: a) Dredging and Spoiling, b) Trace Elements, c) Synthetic organics and petroleum hydrocarbons, d) Sediments and Water, and e) Invertebrate Ecology and Physiology. The arrangement of references is alphabetical by author within each category in each section. Each annotated reference is numbered and a subject index to the annotated section is included.

A number of literature reviews and bibliographies have been written in the last few years in an attempt to elucidate various aspects of dredging and spoiling and/or availability due to these operations. Some of these works are listed at the beginning of the annotated section.

SOME LITERATURE REVIEWS AND BIBLIOGRAPHIES

RELATING TO POLLUTANT AVAILABILITY

Eisler, R.

1973. Annotated bibliography on biological effects of metals in aquatic environments. U.S. Environmental Protection Agency, Ecological Research Series, EPA-R3-73-007. 287 pp.

This is a selection of 567 annotated references on biological effects of metals to aquatic organisms, indexed by taxa and metal. Preference was given to articles on toxicity of heavy metals to marine, estuarine, and anadromous species.

JBF Scientific Corporation.

1973. Interaction of heavy metals with sulfur compounds in aquatic environments and in dredged material. Prepared for: U.S. Army Corps of Engineers Contract No. DACW33-74-M-0191. 14 pp.

This report is a literature review of sulfur compounds, both naturally occurring and intentionally added, and their effect on the disposal of dredged materials containing heavy metals. The areas concerned include metal-binding action of natural sediments, heavy metal contamination, the metal-binding action of organic and inorganic sulfur compounds, and the application of these compounds to complexing of heavy metals in dredged materials.

Lee, G.F.

1970. Factors affecting the transfer of materials between water and sediments. Literature Review No. 1, Eutrophication Information Program, Water Resources Center, University of Wisconsin, Madison, WI. 35 pp.

This is a review of factors which may affect sediment-water exchange. The exchange reactions have been divided into physical factors, which are mainly hydrodynamic effects; biological factors, such as nitrogen release by bacteria, photosynthesis, and respiration, and reduction of sulfates to sulfides; and chemical factors, including chemical characteristics of the water and sediment and the kinds of reactions leading to exchange (i.e. protolysis, precipitation, solubilization, complexation, oxidation-reduction, and sorption). The lack of existing knowledge of the factors influencing exchange reactions is emphasized.

Lee, G.F. and R.H. Plumb.

1974. Literature review on research study for the development of dredged material disposal criteria. Prepared for: U.S. Army Engineer Waterways Experiment Station, Corps of Engineers, Contract No. DACW39-74-C-0024. 145 pp.

This report is a literature review and discussion of the factors influencing the release of chemical contaminants from natural water sediments with emphasis on review of the Elutriate Test.

Lopez, J.M.

1974. Literature Review: Environmental significance of metal contaminants related to dredging and dredged sediments disposal. Manuscript. 24 pp.

The major area of concern in this review is the release of metals from solids and interstitial water resulting from dredging and dredge disposal. Chemical and biological factors are discussed, and metals release studies are reviewed. Emphasis is placed on the EPA proposed water quality criteria and the problems of application to dredged material.

May, E.B.

1973. Environmental effects of hydraulic dredging in estuaries. Alabama Mar. Resour. Bull. No. 9. 85 pp.

This report contains a historical review of dredge studies from 1938-1973. The author concludes that the chemical, physical, and biological effects of dredging have not been properly emphasized and many studies are inconsistent and misleading.

Morton, J.W.

1973. A selected bibliography on dredging and the disposal of dredge spoils. Manuscript. 23 pp.

This is a comprehensive bibliography on dredging which includes references concerning the effects of dredging and dredge disposal, alternative disposal practices, current regulations on dredging and spoiling, research needs, and methods used in dredging practices. Also included is a list of other bibliographies on dredging.

Sherk, J.A., Jr. and L.E. Cronin.

1970. The effects of suspended and deposited sediments on estuarine organisms. An annotated bibliography of selected references. N.R.I. Reference No. 70-19. National Resources

Institute, University of Maryland. Chesapeake Biological Laboratory, Solomons, Maryland. 62 pp.

The direct effects of sediments on estuarine organisms is the major concern of this annotated bibliography. It contains 161 references plus a section on other bibliographies pertinent to the same subject.

U.S. Army Corps of Engineers.

1975. Dredge disposal study, San Francisco Bay and Estuary. Appendix D: benthic community study. San Francisco District, San Francisco CA.

A literature review on heavy metal toxicity is included in this report. Emphasis is placed on sources of heavy metals in aquatic systems, especially San Francisco Bay; toxicity of heavy metals to benthic organisms; and accumulation and levels of heavy metals in invertebrates. Tables are included which summarize toxicity and concentration levels in organisms. Some factors influencing availability are also discussed.

U.S. Army Corps of Engineers.

1975. Dredge disposal study, San Francisco Bay and Estuary. Appendix F: Crystalline matrix study. San Francisco District, San Francisco, CA.

This report contains a literature review on the chemistry of metals in sediment and water, and factors influencing adsorption and desorption. Studies of the environmental effects of dredging are reviewed as well as San Francisco Bay dredge studies. Tables comparing levels of metals in sediment, water, and interstitial water are also included.

A. DREDGING AND SPOILING

1. Biggs, R.B.

1968. Environmental effects of overboard spoil disposal. Proc. Amer. Soc. Civ. Engin. J. Sanit. Engin. Div. 94:477-487.

Background information on the natural environment of Chesapeake Bay is presented. Physical and chemical aspects of spoil disposal are considered and a framework for biological studies of this area is established. Five transects were located between Turkey Point and Tolchester near the mouth of the Susquehanna River with 4 to 6 stations along each transect. Samples were obtained at 3 m depth intervals from surface to bottom on a biweekly basis over a year long period. Those parameters measured include seston, temperature, salinity, and wind and current velocity. The dredging and spoiling occurred from October 7 to November 11.

The seston concentrations during the disposal period were lower than concentrations recorded at other times of the year. Nutrients in the water column were measured after the spoil was allowed to settle for 4 hours. Phosphate and nitrogen levels were significantly increased indicating a release of nutrients from the spoil material into the water column.

2. Brehmer, M.L.

1967. A study of the effects of dredging and dredge spoil disposal on the marine environment. Virginia Inst. Mar. Sci., Spec. Sci. Rept. No. 8. 25 pp.

A study was conducted to determine the effects of dredging and spoiling on Chesapeake Bay. Sediment samples were taken and analyzed for grain size, carbon, phosphate, magnesium, sodium, and potassium content as well as pH and Eh both inside and outside the spoil area. Benthic invertebrate surveys were made in spoiling, dredging, and outside areas before and after dredging periods. Spoil deposited in a deep estuarine area was found to have an immediate but not lasting effect on benthic fauna. The overall effects of dredging were found to be minimal when compared to the natural seasonal fluctuation.

3. Brown, C.L. and R. Clark

1968. Observations on dredging and dissolved oxygen in a tidal waterway. Wat. Resour. Res. 4:1381-1384.

Evidence indicated that resuspension of oxidizable bottom sediments in a tidal waterway caused significant reductions in the dissolved oxygen (D.O.) concentration of the water. During dredging, D.O. was reduced between 16-83% below normal.

4. Durant, C.J. and R.J. Reimold.

1972. Effects of estuarine dredging of toxaphene contaminated sediments in Terry Creek, Brunswick, Georgia. Pest. Monit. J. 6:94-96.

Oyster and sediment samples were taken over a 7 month period to determine whether dredging near a toxaphene plant would increase the amounts of the pollutant in the biota and perhaps cause mortality. Oysters were collected from a single station on a weekly basis for 3 months and on a monthly basis for 4. Each sample contained a minimum of 12 oysters. Sediment core samples were taken from 3 stations ranging from .2 - 1.4 miles from the toxaphene plant outfall. All samples were analysed for toxaphene by gas chromatography. No significant increase of toxaphene residues were found in either oysters or sediment during the dredging and resultant run-off period.

5. Flemer, D.A., W.L. Dovel, H.T. Pfitzenmeyer, and D.E. Ritchie, Jr.

1968. Biological effects of spoil disposal in Chesapeake Bay. Proc. Am. Soc. Civ. Eng. J. Sanit. Eng. Div. 94:683-706.

This study had two main objectives: (1) to ascertain any direct gross effects of dredging and spoil disposal on the biota; and (2) to provide background data for laboratory studies on the tolerance of species to silt loads. Five transects were used between Turkey Point and Tolchester near the mouth of the Susquehanna River. Year-round observations were made; however, this study concentrates on the spoil disposal period--October to November, 1966. Biweekly cruises were made, obtaining data on water transparency, chlorophyll "a" concentrations, photosynthesis (for productivity estimates), and zooplankton samples. Surface and bottom samples were also taken for fish eggs and larvae. Benthic samples were taken on a quarterly basis, as well as a series immediately before and after the disposal period. Monthly fish samples were taken, and exposure experiments were employed using cages both in and out of the disposal area.

This area was found to be highly productive, intensively used by a variety of fish and invertebrates, and important as a spawning ground and nursery for many valuable fish. No gross

effects were observed from disposal of fine materials on the microscopic plants and animals, the eggs or larvae, nor the adult fish. Significant loss occurred in some bottom species. Some species were not affected and some began to repopulate soon after the spoiling. Based on the year-round observations, the late fall appears to be the time when dredging and spoil disposal would have the least damaging effect on the biological species of Chesapeake Bay.

6 Gannon, J.E. and A.M. Beeton.

1971. Procedures for determining the effects of dredged sediments on biota-benthos viability and sediment selectivity tests. J. Wat. Poll. Cont. Fed. 43:392-398.

Laboratory experiments were made with the amphipod, Pontoporeia affinis to test the effects of dredged material on benthic organisms. Tests were first run to discover sediment preference in this species. Sediment samples were taken from nine Great Lakes Harbors and the bin of a hopper dredge. It was found that the amphipod preferred the open lake sediments and those from the control harbor (Great Sodus Bay). A second test was conducted in which six specimens each of Pontoporeia were placed in each type of sediment. The greatest mortality occurred in the Rouge River and Indiana Harbor sediments. The dredge hopper sample also resulted in 100% mortality. The lowest mortality occurred in Great Sodus Bay, Green Bay, and Toledo Harbor sediments. In general, sediments from badly polluted harbors were more toxic than those from the outer harbors. The author emphasizes experiments with the use of native species under conditions as natural as possible.

7. Godcharles, M.F.

1971. A study of the effects of a commercial hydraulic clam dredge on benthic communities in estuarine areas. State of Florida Dept. of Natural Resources, Technical Series No. 64. 51 pp.

A Maryland soft-shell escalator clam dredge was used in a systematic sampling program to study its effects on representative bottom types (habitats) in Tampa Bay and to conduct clam exploration in Tampa and Boca Ciega Bays, the Cedar Keys area, and Tarpon Springs vicinity. Six experimental stations established in Tampa Bay were visually inspected and sampled with trynet before dredging and at various intervals after dredging. Benthic plug samples were taken at the final sampling. Sediment samples were also taken to assess textural changes by particle size analyses. Collected fauna were identified, counted, and in most instances, measured.

After more than a year no recolonization of sea grasses, Thalassia testudinum and Syringodium filiforme, occurred in any dredged area. Some regrowth of Caulerpa prolifera was observed 86 days after dredging. No increase of clam set was detected during

the study. Analyses of trynet hauls showed no faunal variations between dredge and control plots at any time after dredging and benthic plug samples revealed marked faunal differences at only one station. Redredging revealed no faunal declines except for a marked decrease in quahogs, Mercenaria campechiensis, at one station.

Dredgehead water jets penetrated the substrate 18 inches and uprooted all vegetation. Dredge tracks remained visible from one to 86 days and some spots remained soft for over 500 days. Two stations showed a decrease of silt/clay particles immediately after dredging but only one showed a sustained decrease.

8. Gordon, R.B.

1974. Dispersion of dredge spoil dumped in near-shore waters. *Estuar. Coast. Mar. Sci.* 2:349-358.

Turbidity measurements show that 99% of non-cohesive spoil of high silt content discharged from a scow in the presence of a tidal stream is transported to the bottom as a high speed, turbulent jet. Lateral spread of the jet is about 30% of the water depth. Impact with the bottom produces an outward spreading density current whose speed and thickness vary such that the Froude number of the flow remains about constant. When 2000 m³ of spoil is discharged in water 20 m deep, the density surge carries less than 18% of the spoil outside a circle of 30-m radius and essentially none beyond about 120 m. The residual turbidity in the water column, which drifts with the tidal stream, contains less than 1% of the material discharged; this settles at the fall velocity of the individual particles.

9. Gustafson, J.F.

1972. Beneficial effects of dredging turbidity. *World Dredging and Marine Construction* 8(13)44-48.

This paper speculates on the ability of suspended sediments to act as scavengers for metal ions in the water. The results of two "pilot" studies were presented--leading the author to the conclusion that turbidity has the beneficial effects of scouring metal ions from the water to the bottom sediments where they remain due to low oxygen levels below the surface and the binding of particles together with mucous formed by bacteria and diatoms. He also concludes that resuspension of sediments does not cause release of metals into the water column.

10. Kaplan, E.H., J.R. Welker, and M.G. Kraus.

1974. Some effects of dredging on populations of macrobenthic organisms. *U.S. Fish Bull.* 72:445-480.

Populations of epi- and infauna were studied from 10 mo. before to 11 mo. after a navigation channel was dredged through a

small, shallow lagoon. A new sampler which penetrated 20-30 cm. into the substratum was used.

Current velocities and sedimentation patterns were changed due to an altered distribution of tidal currents, although flushing time was not appreciably altered.

Values of certain particulate and dissolved nutrients changed after dredging, but no correlation was observed between animal populations and fluctuations in nutrients.

Significant reductions in standing crop figures and species and specimen numbers occurred in both the bay and the dredged channel. Mercenaria mercenaria populations were reduced, but there was no evidence of mass mortality. Recovery of biomass in the channel was affected by sediment composition, but seasonal and sediment type variations were not significant in the bay as a whole.

Goose Creek had a high predredging epi-and infaunal standing crop estimated at 36.83 g/m^2 , but the number of organisms/ m^2 was relatively low, indicating a preponderance of large forms.

Productivity of Goose Creek was calculated at $89.87 \text{ g/m}^2/\text{yr}$ before dredging and $31.18 \text{ g/m}^2/\text{yr}$ after dredging. Productivity figures for the mixed peripheral marsh were calculated and the annual loss due to replacement of 10.87 ha of marsh by spoil areas was estimated at 49,487 kg. Altered land usage patterns tended to fix this loss on a permanent basis.

The unusually profound effects of dredging reported for Goose Creek are attributed to its small size and shallowness.

11. Leathem, W., P. Kinner, D. Maurer, R. Biggs, and W. Treasure. 1973. Effect of spoil disposal on benthic invertebrates. Mar. Poll. Bull. 4:122-125.

Hydraulic dredging and spoil disposal behind the inner breakwater in Delaware Bay has an impact over several kilometres from the site of operations. While dissolved oxygen and the density of animals fell in the areas immediately affected, the total impact of this operation appears to have been small. There may even have been some recruitment of animals to the spoil areas after the operations.

12. May, E.B. 1973. Environmental effects of hydraulic dredging in estuaries. Alabama Mar. Resour. Bull. 9:1-85.

Hydraulic channel and shell dredging and open water spoil disposal have little significant immediate effect on water quality in Alabama estuaries. Almost all of the sediment discharged by dredges settles very rapidly and is transported by gravity along the bottom as a separate flocculated density layer and potentially harmful components of the mud are not dissolved into the water. There is a limited, temporary reduction in benthic organisms in areas affected

by dredging. Spoil piles from channel dredges can indirectly affect the ecology and usefulness of estuaries by interfering with water circulation and altering salinity. The basic hydrological concepts which determine the effects of dredging should be applicable in other areas. Extensive regulations apparently are not necessary to protect water quality in open water dredging situations, but spoil disposal practices from channel dredges must be reconsidered and appropriate new disposal plans developed.

13. Pearce, J.B.

1970. The effects of solid waste disposal on benthic communities in the New York Bight. 12 pp. In, FAO Technical Conference on Marine Pollution and its Effects on Living Resources and Fishing. Rome, Italy, 9-18 December.

A detailed biological census and hydrographic survey of the New York Bight was undertaken to determine what effects the dumping of wastes has had on the marine resources. Areas where large amounts of sewer sludge were dumped were devoid of normal benthic populations. Concentrations of Cr, Cu, and Pb were greatest in these areas, and O₂ levels of the bottom waters were very low in the summer months. The dredge spoil area was similarly affected, with heavy metal concentrations as high or higher than the sludge area. Significant levels of chlorinated hydrocarbons were also found in this area along with a lack of normal fauna. Crustaceans exposed to these sewer sludge and dredge spoils in the laboratory and field developed pathological anomalies.

14. Pfitzenmeyer, H.T.

1970. Project C: Benthos. p. 26-38. In, Gross physical and biological effects of overboard spoil disposal in upper Chesapeake Bay. NRI Special Report No. 3.

A study to determine the effects on the benthos of channel dredging and overboard spoil disposal was conducted in the Upper Chesapeake Bay where the water salinity varied seasonally between 0 and 7 parts per thousand.

Sixty-six species of benthic invertebrates were identified from this area. The majority were soft-bottom, detritus-feeding invertebrates which can live in a turbid environment.

Only three dominant species were found: the isopod Cyathura polita, the amphipod Leptocheirus plumulosus, and the polychaete worm Scolecopleides viridis. Most of the invertebrates showed considerable variation in numbers of individuals and species related to seasonal increases and decreases in salinity.

Following dredging and spoil disposal, a 71% reduction in average number of individuals per .1m² sample occurred in the spoil

area. At the same time, the deeper channel area showed a 50% increase due to a recruitment of the worm, S. viridis. Subsequent sampling indicated the populations in the channel to be very erratic and temporary. Approximately a year and a half after dredging activities, the number of individuals in the disposal area was the same as the surrounding area. The channel area remained lower and about the same magnitude as before dredging.

The species diversity index, which analyzes the community structure of animals, showed a marked reduction in values immediately after dredging and disposal. One year later, recovery had taken place in the disposal area but not in the channel.

After dredging, the average dry weight of biological material per sample decreased $.23 \text{ g/m}^2$ or 25% in the study area. The greatest decrease, 72%, took place in the channel region where the bottom material was removed. A decrease of 64% occurred at the stations within the spoil deposition area. In April 1967, five months after dredging, an average increase in biomass appeared at all stations. Each sampling period through December 1968 showed an increase over the previous period. A maximum standing crop of 6.4 g/m^2 was recorded. The population expansion and rapid growth of the clam, Rangia cuneata, was the major factor contributing to this increase.

The benthic fauna of the Upper Chesapeake Bay is subjected to an unusual amount of environmental stress and, as a result, the populations never attain an equilibrium in species, numbers, or biomass. Although decreases in fauna were evident, recovery began soon after dredging and spoil disposal. Because of seasonal changes, populations of benthic animals are lowest in numbers and biomass during the spring months. Therefore, probably late winter or early spring would be the time of the year when least damage would occur to the benthic populations as a result of dredging activities.

15. Reimold, R.J. and C.J. Durant

1974. Toxaphene content of estuarine fauna and flora before, during, and after dredging toxaphene-contaminated sediments. Pest. Monit. J. 8:44-49.

This paper evaluates toxaphene concentrations in selected estuarine fauna, flora, sediment, and dredge spoil before, during, and after the dredging of Terry Creek, Brunswick, Georgia, in autumn 1972. This is the second effort to widen the channel of the creek, which receives the effluent from a nearby toxaphene-manufacturing plant; a 1971 dredging effort was aborted by the State of Georgia. The current study employed safeguards inspired by the 1971 State action: enclosure of dredge spoil in diked areas of unproductive marshland to prevent runoff, and weekly monitoring of Terry Creek biota and sediment to detect the possible role of toxaphene

in any resulting disturbance to the balance of nature. Monitoring of dredge spoil, fauna, and flora showed toxaphene concentrations to be higher during dredging than before or after. Eastern oysters (Crassostrea virginica), reported to be among the best biological monitors, did not demonstrate large changes in toxaphene content resulting from the dredging. The high toxaphene concentration in oysters ranged between 2.0 and 5.0 ppm. The best indicators evaluated were salt marsh cordgrass (Spartina alterniflora) and mummichog (Fundulus heteroclitus). The highest content found in S. alterniflora was 7.5 ppm; the highest concentration in F. heteroclitus was over 200 ppm.

16. Rose, C.D.

1973. Mortality of market-sized oysters (Crassostrea virginica) in the vicinity of a dredging operation. Chesapeake Sci. 14:135-138.

Various statistical methods were applied to determine the extent of the oyster mortality caused by dredging activities. Stations 1-7 (0-595 meters from the spoil bank) had an average mortality of 57%; whereas Stations 8-14 (595-1510 m) had an average mortality of 17%. Thus sedimentation resulting from dredge operations caused an increased oyster mortality of 40% within 595 meters of the spoil bank. Sediments (2-15 cm thick) commonly covered the majority of oysters from the affected area. Theoretical mortality as a result of dredging was estimated to be about 48%. Also included is an introduction some other works on the effects of dredging on oysters.

17. Servizi, J.A.

1971. A study of sediments from Bellingham harbor as related to marinedisposal. p. 227-248. In, J.N. Nath and L.S. Slotta, eds. Proceedings 1971 Technical Conference on Estuaries of the Pacific Northwest. Circular No. 42. Oregon State University, Corvallis, OR 343 pp.

Sediments from Whatcom Waterway, Bellingham were studied in response to a proposed dredging and disposal program. Laboratory study indicated that two types of sediment were involved. Sediment from the inner harbor consisted primarily of putrefying pulp fibers which exerted a significant oxygen demand, created substantial turbidity, and were toxic to juvenile sockeye salmon because of their hydrogen sulfide content. Various methods of widespread dispersal to dilute the sediment appeared impractical, and it was concluded that land disposal of inner harbor sediment would be necessary to

protect fish stocks. Sediment from the outer harbor was a natural silt, not containing hydrogen sulfide, but exerted an oxygen demand and created a highly turbid mixture which settled very slowly. Because dumping of this sediment at the proposed site could also prove harmful to fisheries, hydraulic dredging and local disposal adjacent to the outer harbor was recommended.

18. U.S. Army Corps of Engineers.
1975. Dredge disposal study, San Francisco Bay,
Appendix A: Main ship channel, San Francisco Bar.
San Francisco District, San Francisco, California.

Studies were conducted to determine the toxicity of dredged sediments, the degree of water column degradation, the area influenced by material dispersion, and the impact of the physical removal and deposition of the sediments on the benthic communities prior, during and after dredging and disposal operations in the main ship channel and bar area of San Francisco Bay. The following conclusions were drawn:

- 1) The material dredged was not polluted by any of the presently recognized criteria and would not affect fauna or habitat as a result of substrate alteration.
- 2) Water quality was not altered with respect to salinity, pH or dissolved oxygen; turbidity increased during both the dredging and the disposal operations.
- 3) The accumulation of dredge material on the southern portion of San Francisco Bar did not exceed two inches during any one release. Subsequent movement took place in the fluid sediment layer and the turbid suspension layer and was influenced by tidal currents on the Bar.
- 4) The organisms inhabiting the Bar were characteristic of those found in a shifting sand environment. Deposition of dredged material to a depth of two or three inches would make little difference to organisms able to survive mass sediment movement associated with winter storms. This assumption was not confirmed by a survey of benthic fauna before and after operations, but it was supported by laboratory experiments with Dendroaster excentricus.

19. U.S. Army Corps of Engineers.

1975. Appendix C: Dredge spoils disposal monitoring, San Pablo Bay. In U.S. Army Corps of Engineers, Dredge Disposal Study, San Francisco Bay and Estuary. U.S. Army Corps of Engineers, San Francisco, California.

An investigation of the effects of dredge spoil disposal was carried out in February, 1974, in San Pablo Bay. Measurements taken included dissolved oxygen, light transmittance, conductivity, pH, temperature, and current speed and direction at selected fixed depths. The main objective was to determine whether the disposal of spoil resulted in a reduction of dissolved oxygen, and if observed, the duration of the reduction. Of six dredge spoil releases, only one instance of lower dissolved oxygen was observed. This drop lasted for only a few minutes but the oxygen level remained slightly reduced for the duration of the measuring period (27 min). The other five disposal releases resulted in either no change or a higher level of dissolved oxygen.

20. Serne, R. S. and W. B. Mercer.
1975. Dredge disposal study, San Francisco Bay and Estuary. Appendix F: Characterization of San Francisco Bay dredge sediments-crystalline matrix study. San Francisco District, San Francisco, California.

The quantity and nature of certain heavy metals (Cd, Cu, Hg, Pb, Zn) that may be released from selected San Francisco Bay sediments was examined as a function of various physical and chemical parameters.

Ten sediment sampling stations in San Francisco Bay were selected to represent the range of sediment types and heavy metal concentrations which are involved in routine maintenance dredging and disposal. The sediment samples were characterized with respect to particle size, mineral content, total sulfide, organic carbon, cation-exchange-capacity, PCB and heavy metal contents.

For three regions a semi-selective chemical extraction procedure was utilized to determine the relative distribution of the heavy metals among the various geochemical phases. The greatest portions of heavy metals were found bound in clay or crystalline lattice-like and organic or sulfide-like sites.

A batch sorption-desorption experiment was performed to determine the fate of sediment-bound heavy metals during simulated dredging activities. It was found that under oxygen-rich conditions, statistically significant higher concentrations of Cd, Cu, Pb, and Zn were found in the water column than were found under oxygen deficient conditions. Higher-salinity waters also caused greater releases of cadmium and zinc than were found at low salinity. Length of agitation time in oxygen-rich waters also affected the magnitude of release of copper, cadmium and zinc.

Mechanisms to explain the results are discussed as are relationships of the results to the environmental effects of dredging and disposal in San Francisco Bay. Elutriate trace metal concentration predictor equations and recommendations for future work are presented.

21. Peddicord, R. K., V. A. McFarland, D. P. Belfiori and T. E. Byrd.
1975. Appendix G: Physical Impact Study. In U.S. Army Corps of Engineers, Dredge Disposal Study, San Francisco Bay and Estuary. U.S. Corps of Engineers, San Francisco District, San Francisco, California.

The impact of the presence of fine mineral particles

in the water column on the macrofauna of San Francisco Bay was examined. A unique laboratory facility providing large aquaria with open, once-through flow of water with the desired suspended solids concentrations was employed. The initial research determined the 200-hour LC_{10} , LC_{20} and LC_{50} of suspended kaolin for 18 species of fish and invertebrates. The more sensitive species, including Mytilus edulis, Crangon nigricauda, Morone saxatilis and Cymatogaster aggregata, were selected for study of the influence of temperature and dissolved oxygen on the 10-day LC_x of suspended bentonite, which approximated the fine sediments of San Francisco Bay in mineralogy and particle size distribution. Increasing suspended bentonite concentrations reduced the survival of all species, and tolerance decreased as dissolved oxygen content was lowered. Tolerance of the invertebrates to suspended bentonite varied inversely with temperature, while the survival of fish increased at higher temperatures. Changes in oxygen consumption of Mytilus edulis and Morone saxatilis were noted with increasing suspended bentonite concentrations. In burial experiments only Mytilus edulis and Synidotea laticauda had significant mortalities within 4 days after rapid deposition of up to 8 cm of bentonite. No species occurring primarily on muddy bottoms was found to be sensitive to high suspended bentonite concentrations. The sensitive species were all fish not intimately associated with the bottom, sandy bottom epifauna or fouling organisms, none of which normally encounter high suspended solids concentrations for extended periods of time. The results indicate that the impact of high suspended solids would be less in winter than in summer due to higher dissolved oxygen levels and lower temperatures.

22. Anderlini, V. C., J. W. Chapman, D. C. Girvin, S. J. McCormick, A. S. Newton and R. W. Risebrough. 1975. Appendix H: Heavy metal uptake study. In U.S. Army Corps of Engineers, Dredge Disposal Study, San Francisco Bay and Estuary. U.S. Army Corps of Engineers, San Francisco District, San Francisco, California.

Dredging operations in Mare Island Strait in northern San Francisco Bay were examined between September, 1973 and May, 1974 to determine whether these operations release toxic heavy metals from the dredged sediments, resulting in elevated concentrations of these metals in adjacent sediments and invertebrate populations.

Concentrations of the metals silver, arsenic, cadmium, copper, mercury, nickel, lead, selenium and zinc were monitored prior to, during, and after two dredging periods. Metal concentrations were measured in sediments, and the native invertebrates Macoma balthica, Neanthes succinea, Ampelisca milleri, Mytilus edulis and

Ischadium demissum. Mytilus edulis transplanted into Mare Island Strait from Tomales Bay, a relatively undisturbed area on the California coast, were also monitored.

Changes in the mean metal concentrations in sediments and invertebrates during the study period were relatively small, considerably less than an order of magnitude.

The two dredging periods coincided with the two heaviest rainfalls of the year, and the resultant freshwater runoff caused significant changes in the salinity and particulate load in Mare Island Strait. It was, therefore, not possible to determine whether changes in metal concentrations at stations within the dredge zone were caused by either dredging or rainfall phenomena. However, changes in metal levels at stations outside of the dredge zone were of comparable magnitude and direction to those exposed to dredging activity.

The data indicate that dredging activity within Mare Island Strait did not significantly affect the concentration of heavy metals in adjacent sediments and in local invertebrate populations.

Monitoring of lead concentrations in suspended particulates and in centrifuged water samples obtained in the vicinity of the dredge area before, during, and after the operations, showed a significant increase in lead concentrations in uncentrifuged water samples and in the suspended particulates during the first dredging period. Comparable changes were not observed during the second dredging period, suggesting that the observed changes resulted from surface runoff.

Uptake and accumulation of the chloride salts of silver, cadmium, copper, lead, and mercury by the clam Macoma balthica was examined in a laboratory study. Exposure to three concentrations of these metals under three salinity regimes, demonstrated uptake above pre-exposure levels and that lower salinities were associated with higher levels of accumulation.

23. Wakeman, T.H.

1974. Mobilization of heavy metals from resuspended sediments. Presented at the American Chemical Society Conference. Atlantic City, New Jersey. September 11.

An experiment was conducted to determine if there are detectable increases of heavy metal levels in the water column during dredging or disposal operations. Water samples were taken from Mare Island Strait and Carquinez Straits both with and without the influence of dredging and spoiling. The data show that the levels of zinc, lead, chromium, and nickel were significantly greater during the period of dredging and disposal operations. Mercury decreased in one case and showed no significant change in all others. These results were not in agreement with other studies which conclude that dredging operations do not result in any massive release of heavy metals. One possible explanation for the results of this study were the unusual climatic conditions which included unusually heavy rains.

24. Wakeman, T.H., P.L. Knutson, and J.F. Sustar.

1974. Effects of hopper dredge disposal on the dissolved oxygen concentration in North San Francisco Bay. 10 pp. In, Proceedings Tenth Annual Conference Marine Technical Society. Washington, D.C. September 23-25.

The dissolved oxygen concentration was monitored during the U.S. Army Corps of Engineers disposal operations at Carquinez Straits and San Pablo Bay disposal sites. These disposal sites are used in conjunction with annual maintenance of the navigation channels in Mare Island Strait and Pinole Shoal, respectively. The objective of the studies was to determine whether local hopper dredge disposal operations result in an observable fluctuation in the dissolved oxygen concentration of the receiving waters.

The disposal operations in Carquinez Straits were monitored during 1973 dredging operations on two days using fixed and moving vessels. Discrete readings were taken with continuous monitoring probes with supplementary water samples analyzed by Winkler Titration procedures. Disposal operations at the San Pablo Bay site were monitored for two days in 1974 at a fixed station utilizing multiparameter probes and water samples. Prior to both dredging operations bottom samples were taken in the channels to characterize the chemical attributes of the sediments which may affect the investigated phenomena. Additionally, sediment samples were taken from the hopper of the dredge prior to disposal for chemical oxygen demand determinations.

The disposal of bottom sediments from a hopper dredge were shown to produce both increases and decreases in the dissolved oxygen concentration in the receiving water. The actual effect of a particular disposal operation on the dissolved oxygen level is dependent on many factors, the two most important factors being the chemical composition of the dredged sediments and the amount of mechanical perturbation associated with dredge operations.

25. Windom, H.L.

1972. Environmental aspects of dredging in estuaries. Proc. Am. Soc. Civ. Engin. J. Waterways, Harb. Coast. Engin. Div. 98: 475-487.

This report is based on results of the initial phase of a comprehensive project designed to evaluate the nature and magnitude of environmental changes resulting from dredging activities in the salt marsh estuarine environment of the southeastern Atlantic coast. The present report deals with two aspects of the project and specific results, and consideration will be limited to: (1) The chemical response of salt marsh sediments to the deposition of dredged materials; and (2) the water quality response to the dredging and deposition of sediment in the salt marsh estuarine environment.

On the basis of preliminary results of water quality studies related to dredging activities, the following tentative conclusions can be made.

In natural and relatively unpolluted areas, dredging has no significant effect on water quality whether diked or undiked confinement techniques are used.

In polluted areas in marine environments, water quality impairment caused by dredging activities does not necessarily bear any simple relation to the composition of the sediments to be dredged.

In order to evaluate the possible effects on water quality of dredging sediments in a particular area, specific information must be obtained in that area depending on its own characteristics. No general criteria can be set up for dredging in marine waters until a significant variety of dredging situations has been studied in order to have broad experience in possible water quality effects due to dredging activities.

The time which the water mixed with the dredged material is allowed to stay in the spoil area will greatly influence the quality of the effluent from the spoil bank.

Dredging of polluted sediments do not necessarily impair water quality in estuarine environments.

B. TRACE ELEMENTS

26. Adema, D.M.M., S.I. DeSwaaf-Mooy, and P. Bais.
1972. Laboratory investigations concerning the influence of copper on mussels (Mytilus edulis). Nederlandse Organisatie voor Toegapast - natuurwetenschappelijk Onderzoek Nieuws (TNO Nieuws) 27:482-487. (Dutch).

The influence of Cu on mussels was investigated in a laboratory test. Toxicity and accumulation were determined for eight different Cu concentrations (280, 140, 80, 45, 25, 17, 12, and 10 $\mu\text{g/l}$) by autoanalyzer in a continuous flow system containing artificial seawater. The Cu concentrations were analyzed in the individual test tanks on a daily basis. The results showed that the animals subjected to 45 $\mu\text{g/l}$ or more died very rapidly. The Cu contents of the mussels varied from 50-90 $\mu\text{g/g}$. At 25 $\mu\text{g/l}$ Cu, the mussels died within 30 days and accumulated 60 $\mu\text{g/g}$. Smaller amounts of Cu showed less mortality and less accumulation. At 10 $\mu\text{g/l}$ there was no mortality and no accumulation (same as controls which contained about 20 $\mu\text{g/g}$) of Cu.

27. Applequist, M.D., A. Katz, and K. K. Turekian.
1972. Distribution of mercury in the sediments of New Haven (Conn.) Harbor. Environ. Sci. Tech. 6: 1123-1124.

Distribution of mercury in sediments of New Haven Harbor indicates that the primary supply is from municipal sewer outfalls in the harbor. There was a direct correlation of highest mercury concentrations with the location of major municipal outfalls.

28. Boothe, P.N. and G.A. Knauer.
1972. The possible importance of fecal material in the biological amplification of trace and heavy metals. Limnol. Oceanogr. 17: 270-274.

The concentrations of As, Cd, Co, Cr, Cu, Fe, Mn, Pb, and Zn in a brown alga Macrocystis pyrifera (Linnaeus), and in fecal material of the crab Pugettia producta (Randall) fed exclusively on M. pyrifera were measured by atomic absorption spectroscopy. In this closed system consisting of a primary producer and a primary consumer, the magnitude of concentration of these elements in the resulting feces of the consumer was determined. Except for Cr and Cd, whose concentrations in the fecal material were less than in the kelp, all other elements showed levels in the feces 2.3 -- 14 times those in the algae. These data strengthen the suggestion that fecal material may play a significant role in the cycling of elements in the marine environment.

29. Boyden, C.R., and M.G. Romeril.
1974. A trace metal problem in pond oyster culture.
Mar. Poll. Bull. 5:74-78.

Crassostrea gigas reared in the cooling waters of Hinkley Point power station were found to have accumulated high concentrations of copper and zinc. Sufficiently high copper levels caused markedly greening of the soft tissues. The following were the probable sources of copper and zinc: the corrosion of the power station condenser tubes; high metal concentrations and heavy silt load within the Severn Estuary; and corrosion of the metal components in the ponds where the oysters were being grown. Preliminary feeding experiments with oysters maintained in differing silt loads and types suggested that particulate bound metal may be important in determining concentrations of metals accumulated by filter feeding in bivalves; however the particulate appears to be dependent on the metal concerned; in other words, the soluble component may have greater importance in many instances. Oysters placed for a period of four months at power station intake accumulated significantly higher concentrations of several metals than those recorded from "clean" environments. Concentrations of 780 and 5000 ppm for copper and zinc respectively were observed. Metal loads in the water and sediments of the Severn Estuary and cooling water from the power station did not appear to be the cause of high metal concentrations of the oysters. The oysters, various invertebrates, algae, sediments, and water were analyzed by atomic absorption spectrophotometry.

30. Braman, R.S. and C.C. Foreback.
1973. Methylated forms of arsenic in the environment.
Science 182: 1247-1249.

Environmental samples were analyzed for arsenate and arsenite ions and the methylarsenic acids in nanogram amounts. Dimethylarsinic acid and methylarsonic acid were found in natural waters, bird eggshells, seashells, and human urine.

31. Brooks, R.R. and M.G. Rumsby.
1965. The biogeochemistry of trace element uptake by some New Zealand bivalves. Limnol. Oceanogr.
10: 521-527.

Spectrographic determinations of Ag, Cd, Cr, Cu, Fe, Mn, Mo, Ni, Pb, Sb, V, and Zn were carried out on three species of New Zealand bivalves. Analyses were performed on the sediment, on the whole animals excluding shells, on the shells, and on the individual dissected organs. All the

elements concerned showed more enrichment in the shellfish than in the marine environment. Conclusions have been reached concerning the geochemical and biochemical processes involved in trace element uptake by the biosphere.

32. Brooks, R.R. and M.G. Rumsby.

1967. Studies on the uptake of cadmium by the oyster, Ostrea sinuata (Lamark). Aust. J. Mar. Freshwat. Res. 15: 53-61.

In order to study dilution and displacement of absorbed ions into filtered sea water and their displacement by absorbed Cd, analyses for Cd, Ag, Mn, Pb, V, and Zn were carried out on freshly killed oysters, specimens maintained for 100 hr in filtered sea water and on a further number treated for the same period with filtered sea water containing 50 ppm Cd. Although a high Cd concentration did not appear to displace other ions, there was some evidence for loss of ions from the visceral mass. Concentrations in oysters equilibrated with sea water containing added Cd indicated that this element was strongly accumulated in the gills, visceral mass, and heart. These findings were confirmed in radiometric experiments involving uptake of ^{115}mCd from sea water containing this radionuclide. These experiments were also used to measure the rate of uptake of Cd which although relatively slow, was appreciable in magnitude. Fractionation factors for the various organs for varying Cd concentrations in sea water showed a very similar pattern for all organs studied and indicated cessation of uptake at Cd concentrations between 40 and 140 ppm. It was concluded that uptake was due to unselective adsorption probably by coordination to organic ligands. This theory was strengthened by radiometric experiments on the oyster heart where protein-bound Cd probably did not exceed 25% of the total.

33. Brown, B. and M. Ahsanullah.

1971. Effect of heavy metals on mortality and growth. Mar. Poll. Bull. 2:182-187.

The acute toxicity of six metal salts (mercuric chloride, cadmium sulphate, ferrous sulphate, lead nitrate, zinc sodium citrate, and copper sodium citrate) were tested on Artemia salina and Ophryotrocha labronica, and the effects of three of these salts on growth rate were determined. O. labronica was found to be very sensitive to mercury, and had an LC_{50} in 0.01 ppm Hg within 30 hrs. The order of toxicity for Artemia with an LC_{50} at 1.0 ppm : Hg, Cu, Cd, Fe, Zn, and Pb. For Ophryotrocha the LC_{50} at 1.0 ppm is : Hg, Cu, Zn, Cd, Fe, and Pb. There was no significant suppression of growth of Artemia at 1.0 ppm Cu, but a significant suppression was obtained with Zn and Pb at 10 ppm, 5ppm, 2.5 ppm, and 1.0 ppm Zn,

and 10 ppm and 5 ppm Pb. Ophryotrocha growth was significantly reduced by Cu solutions of 0.1 and 0.05 ppm. No suppression was obtained by Zn or Pb in this species.

34. Brown, B.E. and R. C. Newell.

1972. The effect of copper and zinc on the metabolism of the mussel Mytilus edulis. Mar. Biol. 16:108-118.

The effects of copper and zinc on the metabolism of the mussel Mytilus edulis (L.) and its component tissues were studied. 500 ppm copper sodium citrate inhibited oxygen consumption of the whole animal and gill tissue, but no similar effect was observed on digestive gland tissue. 500 ppm zinc sodium citrate exerted no effect upon gill or digestive gland respiration, and neither metal salt affected the respiration of homogenates of gill, digestive gland or gonad. Direct observation of gill tissues during exposure to the metals revealed that 500 ppm copper sodium citrate caused inhibition of ciliary activity; exposure of tissues to 2 ppm Cu for 24 h resulted in only partial inhibition of the cilia. It is suggested that metabolic suppression noted in whole animals and gill tissues is due to the inhibition of an energy-consuming process such as ciliary activity rather than interference with respiratory enzyme systems.

35. Bryan, G.W.

1964. Zinc regulation in the lobster Homarus vulgaris. I. Tissue zinc and copper concentrations. J. Mar. Biol. Ass. U.K. 44: 549-563.

Concentrations of Zn have been determined in the tissues of normal lobsters from sea water containing about 5 µg/l. of Zn. The blood usually contains about 6 µg/g of Zn, mostly in the serum, while the soft tissues contain between 10 and 50 µg/g. Long exposure to sea water containing 100 µg/l. of Zn fails to alter the Zn concentrations of the blood, muscle and gonads, but increases the levels in the urine, excretory organs, hepatopancreas and gills. This result suggests that extra Zn is absorbed from high Zn sea water, possibly via the gills, and that two methods by which this is removed from circulation are by urinary excretion and absorption by the hepatopancreas.

Changes in the blood Zn concentration produced by injection are rectified quite rapidly in the same way. Injections failed to change the Zn concentrations of muscle and gonads.

Further experiments showed that Zn can be absorbed rapidly from the stomach and reaches the blood partly at least via the hepatopancreas. Again, this extra Zn is removed from the blood by urinary excretion and by absorption in the hepatopancreas. Eventual removal of Zn in the hepatopancreas probably

takes place via the blood and excretory organs rather than via the gut. The role of the gills and gut in the regulation of the body Zn content has not yet been studied.

36. Bryan, G.W.

1967. The metabolism of Zn and ^{65}Zn in crabs, lobsters, and fresh-water crayfish. P. 1005-1016. In B. Aberg and F.P. Hungate (eds.). Radioecological concentration Processes. Proc. Int. Symp., Stockholm, Sweden, 1966.

Experiments were conducted with the shore crab Carcinus maenas on regulation of Zn and ^{65}Zn . The results are compared to the lobster Homarus vulgaris and the fresh water crayfish Austropotamobius pallipes pallipes.

The crab maintains a comparatively constant Zn content so that the potential equilibrium concentration factor for ^{65}Zn is almost inversely proportional to the sea water Zn content. By processes which involve Zn binding by the blood, plus possible facilitated uptake in low Zn sea water, ^{65}Zn can be absorbed directly from sea water across the body surface (gills). The turnover of Zn is more rapid in high Zn sea water and consequently ^{65}Zn reaches the lower equilibrium concentration factors in high Zn sea water more rapidly.

Absorption of ^{65}Zn and Zn from food is a comparatively rapid and complete process and is likely to be a more important route for the uptake of ^{65}Zn than direct absorption from sea water.

As Zn turnover is more rapid in high Zn sea water, ^{65}Zn will be lost more rapidly into high than into low Zn sea water. However, an even more rapid turnover of Zn can be induced by feeding with inactive food. Absorption of inactive Zn from food increases the rate of loss of Zn and ^{65}Zn across the body surface, and further losses occur in the faeces and urine.

The position in the lobster is generally very similar except that Zn loss in the urine tends to take the place of loss across the body surface.

The impermeability of the fresh water crayfish eliminates ^{65}Zn absorption and loss across the body surface. Therefore, apart from the contamination of the body surface due to adsorption, ^{65}Zn must all be absorbed from food. The hepatopancreas absorbs Zn and ^{65}Zn from food in the stomach and regulates the body Zn content. Removal of Zn and ^{65}Zn occurs in the faeces and cannot be continued unless food is eaten and faeces are produced.

37. Bryan, G.W. and L.G. Hummerstone.

1971. Adaptation of the polychaete Nereis diversicolor to estuarine sediments containing high concentrations of heavy metals. J. Mar. Biol. Ass. U.K. 51:845-863.

The concentration of copper in Nereis diversicolor O.F. Müller is roughly related to the total concentration in the sediment and particularly high concentrations are found where mining pollution occurs. In contrast, the concentration of zinc in Nereis remains remarkably constant despite wide variations in the environment and appears to be accurately regulated.

In worms from different estuaries concentrations of copper have been related to those of the sediments, sediment extracts and interstitial water at different stations, and the influence of salinity and size of animal has also been considered. The relative importance of the absorption of copper via the gut or across the body surface is still uncertain, but uptake over the body surface appears to be important and in high-copper animals from polluted areas much of the metal is deposited in the epidermis of the body wall and in parts of the nephridia.

High-copper animals survive in polluted areas because they have developed a tolerance to the toxic effects of copper which is neither readily lost, nor readily gained by non-tolerant animals. The situation may be similar to that found on old mine dumps where populations of metal-tolerant land-plants have evolved.

38. Bryan, G.W. and L.G. Hummerstone.

1973. Adaptation of the polychaete Nereis diversicolor to estuarine sediments containing high concentrations of Zinc and Cadmium. J. Mar. Biol. Ass. U.K. 53:839-857.

Concentrations of zinc and cadmium in the polychaete Nereis diversicolor O.F. Müller have been compared with those of the sediments in the estuaries of 26 rivers which drain the old metalliferous mining areas of South-West England. Whereas in the sediments concentrations of zinc varied by a factor of 30 from about 100 to 3000 µg/g, concentrations in the worms, after correcting for size, varied by only 2.7 from 130 to 350 µg/g dry weight. Concentrations of cadmium in the sediments varied by a factor of 46 from about 0.2 to 9.3 µg/g and concentrations in the worms were roughly proportional to them and varied by a factor of 45 from 0.08 to 3.6 µg/g. These results suggest that zinc is regulated by the worm whereas cadmium is not.

Experimental work has shown that with increasing concentrations in solution the rate of absorption of cadmium by Nereis increases more rapidly than that of zinc and is more nearly proportional to the external concentration. This helps to explain why, in the field, concentrations of zinc in the worms vary less than those of cadmium, but another reason is that populations from high-zinc sediments are better at regulating zinc than normal populations. In toxicity experiments they are more resistant to zinc than normal worms and this adaptation is explained by a reduced permeability to zinc and probably to more effective excretion.

Some factors which influence the absorption of heavy metals are discussed and the adaptation of Nereis to high concentrations of metals is compared with that in other organisms.

39. Bryan, G.W. and L.G. Hummerstone.

1973. Adaptation of the polychaete Nereis diversicolor to manganese in estuarine sediments. J. Mar. Biol. Ass. U.K. 53:859-872.

Concentrations of manganese have been measured in Nereis diversicolor, interstitial water and sediments from estuaries draining the mineralized areas of South-West England. Results from the field observations have been supported by experiments using ^{54}Mn . Manganese appears to occur in Nereis in two pools, one exchanging slowly and the other more rapidly. The slowly exchanging pool amounts to roughly 10 $\mu\text{g/g}$ on a dry-weight basis and upon this is superimposed the more rapidly exchanging pool, the magnitude of which depends on several factors. The two most important of these are the concentration of manganese in the interstitial water and salinity. With decreasing salinity the concentration factor for manganese increases, and therefore the highest concentrations of manganese are usually found in animals from the least saline areas of an estuary.

40. Calabrese, A. and D.A. Nelson.

1974. Inhibition of embryonic development of the hard clam, Mercenaria mercenaria, by heavy metals. Bull. Environ. Contam. Toxicol. 11:92-97.

Mortality was determined in the embryos of the hard clam, Mercenaria mercenaria, after addition of the metal salts: mercuric chloride, silver nitrate, zinc chloride, nickel chloride, and lead nitrates at ten different concentrations. Hg and Ag were 100% lethal at .0075 ppm and 0.045 ppm. Zn and Ni, although not as toxic, were 100% lethal at 0.25 and 0.60 ppm respectively. Pb was the least toxic metal with 100% mortality at 1.2 ppm. LC 50 values were also determined for all the metals. Comparisons were made to previous studies.

41. Calapaj, G.G. and D. Ongaro.

1971. Selective assumption of Mn-54 by the mollusc Pinna nobilis. Rev. Intern. Oceanogr. Med. 21:125-133.

The assimilation of Mn-54 by the mollusc Pinna nobilis was investigated in a laboratory experiment. The rate of assimilation resulted in an increase with the concentration of the radioisotope in seawater and a decrease following the addition of

stable Mn. Equilibrium conditions are reached within the sixth day with Mn-54 concentrations of 0.5 pc/ml. The effective half-life of Mn-54 in Pinna seems to be shorter than the physical one.

42. Chipman, W.A., T. R. Rice, and T.J. Price.

1958. Uptake and accumulation of radioactive zinc by marine plankton, fish and shellfish. U.S. Fish. Wildl. Serv. Fish. Bull. 135:279-292.

The zinc content of sea-water samples collected from inshore waters along the Atlantic and Gulf of Mexico coasts averaged 10.6 γ (micrograms) per liter and ranged from a trace to 24.56 γ per liter; the higher values were for samples from areas known to receive metal contamination.

There was a seasonal difference in the zinc content of the sea water at Beaufort, N.C., the lower values occurring during the winter months. The monthly averages ranged from 2.8 γ to 14.6 γ per liter. The average of all the observations was 9.6 γ per liter.

That oysters, clams, and scallops contain large amounts of zinc, thousands of times more than the sea water per unit of weight, was confirmed. In these the greatest accumulation is by oysters, less by hard-shell clams, and least by bay scallops.

Radioactive zinc present in the surrounding water is rapidly taken up in great amounts by these shellfish, probably because of the great difference between the zinc content of the water and that in the tissues. Much of the zinc in the mollusks is exchangeable with that of the water.

High concentrations of the Zn ⁶⁵ injected into or taken up by oysters and scallops occur in the gills. Considerable amounts accumulate in the kidney of scallops. There is also accumulation in the hepatopancreas of the shellfish, but only small amounts in the adductor muscle.

The marine diatom, Nitzschia closterium, takes up large amounts of Zn ⁶⁵ when it is present in the sea water. Although the greater part of the zinc of the cells is exchangeable with that of the water, very little accumulated Zn ⁶⁵ leaves the cell when they are resuspended in nonradioactive sea water. This species of marine phytoplankton appears to accumulate considerable amounts of zinc.

Marine fish quickly take zinc into the body from the digestive tract. Apparently much of it is excreted rather promptly. High blood concentrations of Zn ⁶⁵ from feeding of the nuclide to fish were quickly followed by rapid uptake by the kidney, liver, and other internal organs. The Zn ⁶⁵ concentrations of these and the blood very quickly declined after reaching this early peak. The liver had the greatest accumulation. A slow and long-continued accumulation took place in bone, integument, and muscle tissues.

Although there is an immediate loss of accumulated Zn^{65} when marine fish exposed to the nuclide in sea water are returned to flowing non-radioactive water, a small percentage remains with only very slight loss over periods of many days.

43. Cross, F.A., T.W. Duke, and J.N. Willis.

1970. Biogeochemistry of trace elements in a coastal plain estuary: distribution of manganese, iron, and zinc in sediments, water, and polychaetous worms. Chesapeake Sci. 11:221-234.

The distribution of manganese, iron, and zinc in sediments, water, and in six species of polychaetous worms is described for the Newport River estuary near Beaufort, North Carolina. Concentrations of manganese, iron, and zinc in 0.1 N HCl extracts of sediment samples collected monthly from three stations for 2 years varied with element, location, time, and sediment type. At each station, iron was the most abundant element present in the 0.1 N HCl extracts and zinc was the least abundant. The concentrations of all three elements in the sediments decreased in a seaward direction. There were definite temporal fluctuations in the concentration of these metals in the sediment, although no seasonal trend was evident. In addition, muddy sediments contained higher concentrations of trace metals than sandy sediments.

Concentrations of manganese and iron in the water also decreased in a seaward direction, although concentrations of zinc remained relatively constant. Except for zinc at one station, at least 30% of the total amount of manganese, iron, and zinc in unfiltered water samples could not be measured by standard spectrophotometric techniques without prior treatment with acid. The relative order of enrichment of trace metals in sediment when compared with water was zinc > iron > manganese.

Concentrations of these elements varied in six species of polychaetous worms. We did not observe any intraspecific differences in the concentrations of trace metals in three species of worms, however, that were collected from two stations which had substantially different concentrations of trace metal in the sediment. This finding suggests that either these organisms may be able to regulate their trace metal content or that much of the metal associated with the sediment is in a chemical form which is unacceptable to them. The order of enrichment of trace metals in polychaetes relative to sediment and water was zinc > iron > manganese.

44. Cross, R.J.

1973. Organomercurials in the environment. Chemistry and Industry 15:719-721.

This paper discusses the entrance of mercury into the

waterways and its transformation to methylmercury. Various methods for removing mercury from water are discussed and discarded as either too costly or insignificant in effect. It is suggested that a good beginning would be to prevent mercury from entering the system.

45. DeWolf, P., W. Ch. DeKock, and A. Stam.

1972. Field experiments on the influence of copper and mercury in a natural mussel bed. Nederlandse Organisatie voor Toegapast-Natuurwetenschappelijk Onderzoek Nieuws (TNO Nieuws) 27:497-504. (Dutch).

The determination of metal contents in organisms is of little use if the effects of such metals (physiological as well as ecological) cannot be determined at the same time. Physiological effects can probably best be studied in the laboratory, whereas for the study of ecological effects, field tests are the obvious answer.

In order to find out whether it is possible to carry out toxicity tests in the sea, on a limited scale and under more or less controlled conditions, some basins were built around part of a mussel bed on the Balgzand. The basins were flooded by the tides, and the mussel population was exposed to various concentrations of Cu^{++} (300, 30 and 20 $\mu\text{g/l}$) and of Hg^{++} (6 $\mu\text{g/l}$), added to the sea water. It appeared that even low concentrations of Cu^{++} (20 $\mu\text{g/l}$) exert marked, though sometimes delayed, effects on the biocoenosis, resulting in mortality, accumulation of the metals, and decreased production of organic matter.

46. Eisler, R.

1971. Cadmium poisoning in Fundulus heteroclitus (Pices: Cyprinodontidae) and other marine organisms. J. Fish. Res. Bd. Canada 28:1225-1234.

In acute toxicity bioassays with $\text{CdCl}_2 \cdot 2\frac{1}{2}\text{H}_2\text{O}$ at 20 C and 20‰ salinity, the concentrations fatal to 50% of the organisms of various marine species in 96 hr ranged between 0.32 and 55.0 mg/liter Cd^{2+} . The order of sensitivity was: sand shrimp, Crangon septemspinosa (0.32); hermit crab, Pagurus longicarpus (0.32); grass shrimp, Palaemonetes vulgaris (0.42); common starfish, Asterias forbesi (0.82); common soft-shell clam, Mya arenaria (2.2); green crab, Carcinus maenas (4.1); Atlantic oyster drill, Urosalpinx cinerea (6.6); eastern mud snail, Nassarius obsoletus (10.5); sandworm, Nereis virens (11.0); striped killifish, Fundulus majalis (21.0); blue mussel, Mytilus edulis (25.0); sheepshead minnow, Cyprinodon variegatus (50.0); and mummichog, Fundulus heteroclitus (55.0). Mummichogs

were more susceptible to cadmium exposures at 20°C than at 5°C and at 5‰ salinity than at 15, 25, or 35‰. Additional studies with mummichogs clearly demonstrated that 96 hr was not sufficient to adequately evaluate cadmium toxicity to this species. Mummichog whole body cadmium residues determined by atomic absorption provided a useful index of cadmium body burden among fish surviving exposure. However, cadmium residue data from dead mummichogs were of limited worth owing to accumulation after death.

47. Eisler, R., G. E. Zarogian, and R. J. Hennekey.
1972. Cadmium uptake by marine organisms. J. Fish.
Res. Bd. Canada 29:1367-1369.

Adults of mummichog, Fundulus heteroclitus, scallop Aquiptecten irradians, oyster Crassostrea virginica, and subadult lobsters Homarus americanus were immersed for 21 days in flowing sea water containing 10 µg/liter of cadmium as $\text{CdCl}_2 \cdot 2\frac{1}{2} \text{H}_2\text{O}$. Cadmium residues in whole animals and selected tissues were consistently higher in exposed organisms than controls; edible portions of treated lobster (muscle), scallop (adductor muscle), and oyster (whole animal) contained more cadmium per unit wet weight than controls by 25%, 19%, and 352%, respectively.

48. Elderfield, H., L. Thornton, and J.S. Webb.
1971. Heavy metals and oyster culture in Wales. Mar.
Poll. Bull. 2:44-47.

Investigation on the causes for poor larval development and settling of Ostrea edulis L. were carried out at the White Fish Authority's hatchery at Conway, North Wales. Concentrations of 900 and 250 ppm of zinc and lead respectively were recorded in estuarine sediments of Conway Estuary. The mean background of Zn in the water was 0.025 ppm while the tributary streams draining old mine areas contained up to 3.0 ppm. The Zn content of the water near the input pipe of the hatchery was 0.10 ppm. Lead content of water near hatchery was .0075 ppm whereas the background level was .0045 ppm. Water analysis of hatchery water was conducted and metal analysis was determined by atomic absorption spectrometry. Freshwater input of zinc was highest in February and measured up to 0.47 ppm. The greatest proportion of total zinc was in the water rather than in the particulate matter. Copper and lead were comparable in both water and particulates. It was reported that less than 0.1 ppm Zn had minimal effect on larvae; however, at 0.3 ppm larval growth was considerably reduced. Either death or failure of metamorphosis occurred at 0.5 ppm. The investigators estimated that 75% Zn was largely present in an ionic and/or weakly complexed form. Tentative evidence of larvae being affected by

concentrations of zinc in the hatchery water was inferred by the authors. Homogenized soft parts of adult oysters contained up to 2500-3000 ppm/ dry wt. Zn.

49. Graham, D.L.

1972. Trace metal levels in intertidal mollusks of California. Veliger 14:365-372.

Determinations of silver, cadmium, chromium, copper, manganese, lead, and zinc were made by atomic absorption spectrophotometry on 7 species of Mollusca in the genera Acmaea, Tegula, Thais (gastropods), and Mytilus, Protothaca, and Tapes (bivalves). Eleven regions along the California coast from San Francisco Bay to Los Angeles were included. Separate analyses were performed on shells and soft portions of whole bodies. Highest levels, above 900 ppm lead in bodies of Acmaea digitalis, 570 ppm copper and 1700 ppm zinc in bodies of Thais emarginata, have raised the question of causes for these unusual concentrations.

50. Halcrow, W., D.W. MacKay, and I. Thornton.

1973. The distribution of trace metals and fauna in the Firth of Clyde in relation to the disposal of sewage sludge. J. Mar. Biol. Ass. U.K. 53:721-739.

Trace metals have been determined in the sediments, waters and fauna of a sewage-sludge disposal area in the Firth of Clyde. Enhanced values of organic carbon and several metals were found in the sediments of the immediate deposit area. The results are discussed in relation to local background variations and compared with data from the Solway Firth. Trace-metal concentrations in the waters of the deposit area were higher than those from further off shore in the Clyde.

Epifaunal species showed rather erratic variation in trace metal content, unrelated to total or readily extractable trace metals in the underlying sediment. The trace-metal content of demersal fish species was not significantly different from figures reported for elsewhere in the United Kingdom.

The distributions of some in-faunal and epifaunal species in the area are described. It is concluded that the effects of sewage sludge disposal in this area are local, gross changes being limited to an area of about 20 km² of sea-bed. However, little is known of the overall effects of toxic wastes particularly at threshold levels, and further monitoring is required.

51. Hallberg, R.O.

1974. Metal distribution along a profile of an intertidal area. Estuar. Coast. Mar. Sci. 2:153-170.

The sediment from an intertidal area has been investigated with regard to the vertical and areal distribution of the following acid-soluble metals: Na, K, Mg, Ca, Fe, Co, Ni, Cu, Ag and Pb. In addition to the metals, granulometric properties, content of organic matter, carbonate concentrations and Eh are measured. The metals are accumulated along the margins of the area and show a high correlation with organic matter and the finest sediment fraction. The interstitial water is diluted with infiltrated rain water. The higher level of this water, of the marsh area compared with that of the tidal flat area, gives rise to an upward percolation in the sediment of the tidal flat area. The metal interrelationships and concentrations are determined by processes in the uppermost oxidized layer and only to a minor extent by reducing processes in the lower hydrogen sulphide-rich layers.

52. Halstead, B.W.

1970. Toxicity of marine organisms caused by pollutants. 21 pp. In FAO Technical conference on Marine Pollution and Its effects on Living Resources and Fishing. Rome, Italy. 9-18 December.

This is a review of toxicity in marine organisms including naturally occurring biotoxins, heavy metals, petroleum, petrochemicals, and pesticides. Emphasis is placed on food resources and the effects of man-caused pollution. Further studies are recommended in these areas.

53. Hobdin, D.J.

1967. Iron metabolism in Mytilus edulis. I. Variations in total content and distribution. J. Mar. Biol. Ass. U. K. 47:597-606.

The iron content of fresh Mytilus edulis L. from Southampton is usually 20-40 $\mu\text{g/g}$ wet weight after the animals have eliminated their gut contents. In spring some animals have much higher iron contents, sometimes in excess of 100 $\mu\text{g/g}$. The much higher values reported by some authors are probably erroneous.

Prolonged starvation in sea water of low iron content will not reduce the mean iron content of the animals below 20-25 $\mu\text{g/g}$. This represents a permanent store. Higher values are produced by a temporary store that is fairly rapidly lost on starvation.

The highest iron concentrations are usually in the digestive gland, which contains the major part of the temporary store, much of which can be regarded as particles being subjected to the digestive processes.

Only iron in the temporary store could be demonstrated by histochemical techniques.

54. Ireland, M.P.

1974. Variations in the zinc, copper, manganese, and lead content of Balanus balanoides in Cardigan Bay, Wales. Environ. Poll. 1:65-75.

This is a study of the seasonal variation of Zn, Cu, Mn, and Pb in Balanus balanoides. The results of this study (and others) indicates that for Zn and Cu lower metal concentrations are found in summer months. The hypothesis is that this is due to phytoplankton numbers increasing, causing lower amounts of metals per phytoplankton (Balanus is a filter feeder). Highest concentrations were found in November and March. The highest concentration of Mn was found in June. This was not explained.

55. Jensen, S. and A. Jernelöv.

1969. Biological methylation of mercury in aquatic organisms. Nature 223:753-754.

Experiments were performed to investigate the methylation of mercury by bottom sediments and rotten fish in fresh water. Mono and dimethyl mercury were found to be produced in both the sediments and fish. This may be significant in the uptake and distribution in fish and the mobilization of deposits from the sediments into the environment.

56. Keckes, S. and J.K. Miettinen.

1970. Mercury as a marine pollutant. 34 pp. In FAO Technical Conference on Marine Pollution and Its Effects on Living Resources and Fishing. Rome, Italy. 9-8. December 1970.

This is a review of mercury in the marine environment including its chemical properties, occurrence and distribution, circulation and transformation, artificial sources, analytical techniques, and biological effects.

57. Leatherland, T.M. and J.D. Burton.

1974. The occurrence of some trace metals in coastal organisms with particular reference to the Solent region. J. Mar. Biol. Ass. U.K. 54:457-468.

Measurements have been made, by neutron activation analysis, of arsenic, cadmium, antimony, mercury and zinc in organisms and bottom muds, mainly from Southampton Water and the Solent. The values are on a dry weight basis and refer to whole organism soft tissues unless otherwise stated.

Concentrations of mercury in algae ranged from 0.07 to 0.22 ppm and those in animal tissues from 0.06 to 1.9 ppm. The distribution of the element in various tissues of Mercenaria mercenaria was rather uniform, except that the concentration in the gills was about 2.5 times that in the whole organism. Variations amongst organisms from different parts of the Solent area and comparisons with other regions suggested some possible influences of local waste discharge, but there was no evidence of major effects. Anoxic muds contained higher concentrations of mercury than occurred in unreduced surface layers, suggesting mobility in interstitial waters of the uppermost zone and fixation under reducing conditions.

The average concentrations of cadmium in algae and fish from substantially unpolluted waters were 0.3 and 0.03 ppm, respectively, while the range for marine Gastropod and Lamellibranch molluscs was from 0.3 to 21 ppm. Increased concentrations, by about two orders of magnitude, were found in Fucus serratus and Littorina littoralis from the Severn Estuary. Bottom muds from areas of low pollution contained, on average, 0.6 ppm.

Concentrations of arsenic in molluscs were usually in the range from 3 to 24 ppm, while fish muscle contained on average 5 ppm. Sepia officinalis contained 73 ppm in mantle tissue; in both Sepia and Mercenaria concentrations were higher in the gills than in mantle tissue. Relatively high concentrations occurred in Tealia (72 ppm) and in most of the algae examined (11-54 ppm). Antimony occurred usually in the range from 0.01 to 0.1 ppm, with low values in fish muscle; concentrations averaging 0.2 ppm were found in two ascidian species. The average concentrations of arsenic and antimony in bottom muds from Southampton Water were 14 and 1.0 ppm respectively.

58. LeBlank, P.J. and A.L. Jackson.

1973. Arsenic in marine fish and invertebrates.
Mar. Poll. Bull. 4:88-90.

An environmental survey has been undertaken on the Pacific coast of Canada to document natural background levels of a number of heavy metals, including arsenic, in various species of marine fish and invertebrates. The majority of fish studied from four different sampling locations near a proposed mine site contained arsenic with levels ranging from less than 0.4 to 37.8 mg/kg (wet weight). Crabs were found to contain the highest level of arsenic of all marine organisms tested. Results indicate that arsenic accumulates in marine fish and invertebrates and may possibly be a micronutrient.

59. Lewis, A.G., P.H. Whitfield, and A. Ramnarine.

1972. Some particulate and soluble agents affecting the relationship between metal toxicity and organism

survival in the calanoid copepod Euchaeta japonica.
Mar. Biol. 17:215-221.

Particulate and water-soluble agents were tested to determine their ability to affect the relationship between metal toxicity and the survival of Euchaeta japonica (Copepoda, Calanoida). Clay minerals and diatoms were two types of particles capable of affecting this relationship. Ascorbic acid, sewage effluent, and water extracts of humic acid and two types of soils exhibited the same capability. The ability of the water-soluble agents was compared with that of a known chelating agent in an attempt to quantify the activity of the agents.

60. Lowman, F.G., T.R. Rice, and F.A. Richards.
1971. Accumulation and redistribution of radio-nuclides by marine organisms. P.161-199. In,
Radioactivity in the Marine Environment,
Washington, D.C., National Academy of Sciences.

A major fraction of the mass and surface area of the biota in the sea is provided by the lower trophic levels of food webs. Biological transport of some trace elements and radionuclides has been attributed to these organisms. One reason is that they are capable of concentrating several elements to levels much greater than their concentration levels in the water; in addition, many zooplankton undergo vertical migration and produce detritus in the form of fecal pellets, moults, and carcasses, which sink because of the influence of gravity. Elements that may be significantly concentrated by marine organisms include structural, catalytic, and heavy divalent elements; heavy halogens; and elements easily hydrolyzed at seawater pH. The amounts of the nutrient elements--phosphate, nitrate, and silicate--increase significantly with increased depth in the sea, in contrast to the amounts of the conservative elements, which change only with salinity.

The heavy divalent ions -- barium and lead, the rare earths, yttrium, scandium, and silver--follow the distribution patterns of the nutrient elements to varying degrees.

Considerations related to biomass, feeding rates, conversion efficiencies, and migratory habits of zooplankton, as well as the chemical characteristics of the elements of interest, suggest that the major downward transport of these elements and radionuclides is effected through the influence of gravity on fecal pellets, moults, and carcasses, with direct biological transport accounting for 10 percent or less of the total movement toward the bottom of the sea.

In estuarine and other nearshore marine regions, the bottom sediments are close to the sites of photosynthesis and to the sites of the introduction of fallout and terrestrial additions of radionuclides. In these marine regions, the bottom sediments and their associated epiphyton often significantly influence the distribution of added radionuclides. In areas containing large populations of sessile filter feeders, these organisms may exert profound effects upon the rates of sedimentation of added trace elements and radionuclides.

61. Majori, L. and F. Petronio.

1973. Marine pollution by metals and their accumulation by biological indicators (accumulation factor). Rev. Intern. Oceanogr. Med. 31-32:55-90.

Laboratory experiments were conducted on the accumulation of cadmium, copper, lead, and mercury by Mytilus galloprovinciales in the form of bivalent ions in seawater at concentrations of 5-500 ppb. It was found that when a certain minimum pollution threshold is exceeded, the initial accumulation rate is proportional to the added pollutant.

An apparent equilibrium accumulation is reached which grows linearly and sometimes proportionally with the pollution level.

The subsequent depletion rate in clean water is always lower than accumulation and seems to be negatively affected by the previous pollution level of mussel and of water and by the pollution duration.

Within these limits of pollution, an univocal simplified correlation between water pollution, mussel accumulation and time can be found when the hypothesis of an equilibrium chemical reaction between some components of the mussel and the metal is set forward.

It is possible to account for the phenomena connected with the accumulation stage and to draw a new parameter : the accumulation factor AF which is more representative to pollution than the concentration factor CF and thereby more useful for quantifying the affinity of metals for biological tissues.

62. McFarren, E.F., J.E. Campbell, and J.B. Engle.

1961. The occurrence of copper and zinc in shellfish. p. 229-234. In Proc. 4th Shellfish Sanitation Workshop. U.S. Dept. HEW.

Polarographic analyses for copper and zinc were conducted on oysters (presumably the American oyster Crassostrea virginica from Chesapeake Bay). Zinc content of 3200-4000 mg/kg (ppm) were found near Baltimore and Annapolis, whereas, in the lower part near Cape Charles, the animals contained around 310-330 ppm. Oysters

from Shelter Island off Long Island had intermediate zinc content around 920-1900 ppm. Oysters contained 10-20 and 30-40 times as much copper and zinc respectively as clams and about 200 and 90 times as much copper and zinc respectively as mussels. Oysters high in zinc are also high in copper, although the converse is not true. Blue-green oysters may be indicative of a high copper content but not necessarily a comparatively high zinc content. Little copper or zinc was found in the liquor after the animal was thawed and the metal content of the meat appeared to increase as the liquor was lost. Copper and zinc in oysters were found roughly as the following: 26.5% mantle, 25.7% gills, 21.5% liver-stomach, 4.6% palps, 4.4% adductor muscle, and 17.5% in the remaining tissue. The investigators conclude that the copper and zinc content is related to the feeding habits of the animal since there was no direct relationship between zinc content of the water and that of the oyster. Oysters could acquire zinc from algae which concentrate zinc very effectively. Toxicity tests of zinc extract of oysters to mice were also conducted. Their results are listed but the conclusions and interpretations of the data were not clear.

63. Morris, O.P., and G. Russell.

1973. Effect of chelation on toxicity of copper.
Mar. Poll. Bull. 4:159-160.

The effect of chelation on the toxicity of copper chloride to brown alga Ectocarpus siliculosus was significant. Growth rate of the alga dropped very rapidly with increasing concentration and no increase in plant volume occurred at 0.45 mg/l. The presence of EDTA increased this concentration up to 0.85 mg/l where the alga continued to grow. It was calculated that reduction in ionic copper concentrations achieved by complex formation was 1 Cu^{++} ion to 1.0-1.5 molecules of EDTA. There was some reduction in the toxicity of copper as the result of chelation of organic material in the soil extract media. A formula was presented that calculated that the soil extract was more effective as a buffer than EDTA.

64. Olson, K.R. and R.C. Harrel.

1973. Effect of salinity on acute toxicity of mercury, copper, and chromium for Rangia cuneata (Pelecypoda, Mactridae). Contrib. Mar. Sci. 17:9-13.

The 48, 72, and 96 hour median tolerance limits (TL_m) for mercury, copper, and chromium were determined for Rangia cuneata at salinities of less than 1 ppt, 5.5 ppt, and 22 ppt. All of the ions were more toxic in freshwater than in more saline waters. Copper and chromium were most toxic; less than 1 ppm was required for a 48 hour TL_m . At 22 ppt salinity all salts required more than 10 ppm for a 48 hour TL_m . 209

65. Peden, J.D., J. H. Crothers, C.E. Waterfall, and J. Beasley.
1974. Heavy metals in Somerset marine organisms. Mar.
Poll. Bull. 4:7-9.

Limpets (Patella vulgata) collected from near low-tide mark contained more cadmium than individuals living in a higher situation. Higher concentrations of Cd were found in larger limpets, and the shells contained small quantities of the metal. There was a reciprocal relationship between concentrations of Zn and Cd in the limpet. Desorption of Cd and Zn were carried out using the limpet, crab (Carcinus maenas L.), and dog whelks (Nucella lapillus L.).

C. maenas, herring gulls (Larus argentatus Pont.), and N. lapillus were analyzed to determine if Cd was transmitted through a food chain since the three animals were considered predators of the limpet. The crab had higher concentrations than C. pagurus which have similar feeding habits. In addition Nucella had considerably higher concentrations than the controls. Analysis of fish and shrimp was undertaken and their concentrations did not approach levels found in the limpets and their predators. All analysis was done using the atomic absorption spectrophotometer.

66. Pentreath, R.J.
1973. The accumulation from water of ^{65}Zn , ^{54}Mn , ^{58}Co , and ^{59}Fe by the mussel, Mytilus edulis.
J. Mar. Biol. Ass. U.K. 53:127-143.

The accumulation from sea water of ^{65}Zn , ^{54}Mn , ^{59}Fe and ^{58}Co by the mussel, Mytilus edulis, has been studied in relation to the stable element levels of these isotopes both in the sea water and in individual tissues. For all four radio-nuclides the greatest accumulation occurred in the stomach and digestive gland samples and further localization of ^{65}Zn and ^{59}Fe was demonstrated by autoradiography. As the animals were starved during the accumulation period the loss of stable elements by individual tissues was also followed. Again the most notable effect occurred in the digestive gland tissues with the exception of a large loss of iron by the foot. Autoradiography showed that after two weeks accumulation ^{59}Fe occurs in large clusters in the foot, notable in the byssus gland area. These clusters disappear after a further two week period and may thus be secreted into new byssus threads.

The accumulation of nuclides was examined using a single exponential model and values obtained for flux rates, biological half times and asymptotic values were compared with the stable element concentration factors. An analysis of parameters of exchange of nuclides in individual tissues with the water was further examined using the Kendall coefficient of concordance which demonstrated that the highest exchange occurs in the order of stomach and digestive gland > gill > foot >

mantle> gonad> adductor. The application of the Friedman test of two-way analysis of variance indicated that this order obtains for all four nuclides studied, despite the fact that in the aquaria used zinc and cobalt were largely soluble; that manganese was partly in the particulate form with the radio-nuclide used; and that iron was largely particulate in both stable and active forms. There is an indication that as well as accumulating nuclides via particulate matter in suspension the mucus itself is capable of sequestering them, even though they are in the soluble form, and may even preferentially accumulate soluble forms. The actual role of water in the accumulation of the nuclides studied appears to be relatively minor compared with that of food accumulation as estimated by difference from the calculated stable element values.

67. Phelps, D.K.

1967. Partitioning of the stable elements Fe, Zn, Sc, and Sm within a benthic community, Anasco Bay, Puerto Rico. P. 721-734. In B. Aberg and F.P. Hungate (eds.) 1966 Radioecological Concentration Processes. Proc. Int. Symp., Stockholm, Sweden.

Polychaetes were collected from two stations and metal concentrations in the two different substrates were compared with the concentrations in the polychaetes. In general, metals partitioning was divided between feeding types. Selective deposit feeders concentrated iron to the exclusion of zinc, and non-selective deposit feeders concentrated zinc to the exclusion of iron. Since selective types feed mainly at the sediment - water interface, they are more likely to accumulate freshly settled particulate or colloidal iron. Non-selective subsurface feeders are exposed to zinc in the reduced form and are more likely to concentrate this metal.

68. Phelps, D.K.

1969. Trace element composition of inshore and off shore benthic populations. P. 509-526. In D. J. Nelson and J.C. Evans (eds). Symposium on Radioecology. Proc. Second Nat. Symp. Ann Arbor, Mich. 1967.

In this study, it was found that non-selective deposit feeders tend to accumulate more zinc than selective deposit feeders, and filter feeders tend to accumulate the most. Partitioning also occurred between non-selective and selective feeders with iron preferentially taken up by the former and zinc by the latter. This was attributed to the difference in feeding habits between the two. Since selective types feed mostly at the sediment-water interface on freshly settled particulates, they are most likely to ingest particulate or colloidal iron. Zinc is more available to subsurface

(non-selective) feeders because of the tendency for zinc to be reduced from the sediments under anaerobic conditions.

69. Preston, A., D.F. Jefferies, J.W. R. Dutton, B.R. Harvey, and A. K. Steele.
1972. British Isles coastal waters: the concentrations of selected heavy metals in sea water, suspended matter, and biological indicators-a pilot survey. Environ. Poll. 3: 69-82.

Data from British Isles coastal waters, on the concentrations of selected metals in sea water and biological indicators, have been obtained by analytical techniques based on atomic absorption spectrophotometry. The results show that there are some areas where significant contamination exists, and the east Irish Sea appears to have the highest concentrations of most metals. However, data in most regions indicate that the concentrations of the metals examined are not significantly higher than those in the open Atlantic Ocean adjacent to the British Isles. The sampling of seaweeds strongly suggests that concentrations of most metals, including those in polluted areas have changed little over the ten years up to 1970; the concentrations of cadmium may well, on average, have fallen over this period. In the Irish Sea, where the most detailed examination has been made, there is a rapid decrease in sea water concentrations from the shoreline to offshore, and, in general, the proportions of a metal associated with suspended matter remain fairly constant with respect to variations of total concentrations in either time or space.

70. Preston, A.
1973. Cadmium in the marine environment of the United Kingdom. Mar. Poll. Bull. 4:105-107.

This is a general review of cadmium in the marine environment : input, distribution, seawater concentrations, concentrations in biota and sediments and concentration factors with particular reference to British waters. Suggestions are given about the direction of future cadmium studies.

- 71 Preston, A.
1973. Heavy metals in British waters. Nature 242 : 95-97.

A survey of metals (Cd, Cu, Fe, Mn, Ni, Pb, Zn) in seawater and organisms was undertaken around the British Isles. Several interesting findings were made. First, contamination is restricted to a few areas, mainly, but not always, linked to

industrial development. Second, such contamination does not spread very far offshore. Third, the amount of contamination has changed very little in the last 10 years. The only exception to this is Cd whose concentration appears to have decreased during this time.

72. Reish, D.J., F. Piltz, and J.M. Martin.

1974. Induction of abnormal polychaete larvae by heavy metals. Mar. Poll. Bull. 5:125-126.

A little known observation that exposure to sublethal concentrations of detergent cause fatal abnormalities in the second generation of a polychaete has been followed up and similar abnormalities have been observed in the first or second generation of larvae exposed to sublethal concentrations of copper and zinc. Few toxicity tests extend beyond the responses of the exposed individuals. The discovery of a delayed reaction to sublethal concentrations of common contaminants of the sea has unknown but potentially important ecological implications.

73. Renfro, W.C.

1973. Transfer of ^{65}Zn from sediments by marine polychaete worms. Mar. Biol. 21:305-316.

Silty marine sediments spiked with ^{65}Zn lose only small fractions of their radioactivity when exposed to slowly flowing seawater for several weeks. However, polychaete worms (Nereis diversicolor), burrowing through the sediment, cause ^{65}Zn losses 3 to 7 times higher than in sediment without worms. Long term experiments on the uptake and loss of ^{65}Zn by the polychaete Hermione hystrix indicate that 60 or more days exposure are required for this worm to approach steady state with ^{65}Zn in the sediment. Biological half-life estimates for ^{65}Zn accumulated from sediment by H. hystrix, are extremely variable (52 to 197 days), depending on the loss-time interval chosen for the calculation. Following 5 days exposure to 16 cm³ of radioactive sediment, N. diversicolor individuals contained an average of 0.2% of the total ^{65}Zn in the sediment. When these worms were transferred to non-radioactive sediment, estimates of biological half-life for ^{65}Zn averaged 14 to 17 days during the loss period Day 3 to Day 15. Based on these experimental results, it is estimated that a population of N. diversicolor could cause an annual loss of 3% or more of the ^{65}Zn in the upper 2 cm of the sediment of a hypothetical radioactive estuary.

74. Rothstein, A.

1959. Cell membrane as site of action of heavy metals. Fed. Proc. 18 : 1026-1038.

Experiments were selected from the literature for a discussion of membranes as the site of heavy metal damage. Some of the conclusions were:

Reactions of heavy metals within the membrane are relatively rapid and reversible, whereas those occurring within the cell show a time lag.

If known functions of the cell membrane are disturbed, then the metal is presumably acting on the membrane.

If the chemical interactions or the resultant physiological effects of the metal are responsive to extracellular rather than intracellular substances, then it may usually be presumed that the interaction is at the cell membrane.

75. Schulz-Baldes, M.

1973. The common mussel Mytilus edulis as indicator for the lead concentration in the Weser Estuary and the German Bight. Mar. Biol. 21:98-102. (German)

M. edulis was collected at 13 stations in the Weser Estuary and the German Bight in three size categories: 14-16 mm, 21-23 mm, and 35-40 mm shell length. These were analyzed separately for Pb concentrations by flameless atomic absorption spectrophotometry. Concentrations of Pb were higher in the Weser Estuary than the German Bight, ranging from 6.4 µg/g to 1.9 µg/g respectively.

Within stations the concentration in small mussels was significantly higher than in larger ones. Average Pb concentrations in the size groups were as follows.

length	14-16 mm	21-23 mm	35-40 mm
ave. dry wt.	12 mg	37 mg	139 mg
ave. Pb conc.	6.81 µg/g	5.68 µg/g	4.95 µg/g

Pb concentrations were highest in the kidney, intestine, and adductor muscle and lowest in the foot, gills, and mantle. It was concluded that M. edulis is highly suitable as an indicator organism for Pb.

76. Tabata, K.

1969. Studies on the toxicity of heavy metals to aquatic animals and factors to decrease toxicity - I. On the formation and the toxicity of precipitate of heavy metals. Bull. Tokai Reg. Fish. Res. Lab. 58:203-214.

Experiments were conducted on the solubility of Zn^{2+} , the factors controlling it, and the toxicity of five precipitates of Zn to marine organisms. The toxicity of Zn precipitates were negligible to Japanese killfish, goby, and Daphnia. Mytilus

edulis, however, were affected by the precipitate. This was assumed to be related to the filter feeding habits of this organism. The experiment solubility of Zn^{2+} in seawater was not found to be regulated by pH values and was larger than theoretical values. Dissolved Sn^{4+} , Cr^{3+} , Al^{3+} , and Fe^{3+} within the pH range shown in water quality criteria were not toxic to the aquatic organisms.

77. Turekian, K.K.

1974. Heavy metals in estuarine systems. *Oceanus* 18:32-33.

This is a short summary of the mechanisms of heavy metal transport in an estuarine system and the subsequent activities of these metals. He concludes that once heavy metals are deposited on the estuarine floor, they are unlikely to be released from the sediments. The metals that are released become adsorbed to particles and are redeposited on the bottom. Strong physical perturbations are the only means of transport.

78. Valiela, I., M.D. Banus, and J.M. Teal.

1974. Response of salt marsh bivalves to enrichment with metal-containing sewage sludge and retention of lead, zinc, and cadmium by marsh sediment. *Environ. Poll.* 7:149-157.

Growth in Mercenaria mercenaria and Crassostrea virginica, bivalves found in tidal creeks in salt marshes on the coast of the NW Atlantic, was not affected by experimental additions of metal-containing sewage sludge and urea fertilisers to salt marsh plots. Modiolus demissus, a mussel inhabiting the marsh surface itself, did grow better under the same fertiliser treatments. All three species of shellfish showed no increases in lead or zinc contents, but all showed increased cadmium contents related to the sludge fertiliser treatments. Increases in zinc, and particularly in cadmium, but not lead, were detected in the creek bottom detritus downstream from the plots. The surface sediment of the marsh plots shows significant accumulations of all three metals. The calculation of input-output budgets shows that lead was trapped in the sediments with virtually no losses to deeper waters. Zinc and cadmium also accumulated in the sediments but there is some transport away from the salt marsh surface, especially in the case of cadmium.

79. Ünlü, M.Y., M. Heyraud, and S. Keckes.

1970. Mercury as a hydrosphere pollutant. I. accumulation and excretion of $^{203}\text{Hg Cl}_2$ in Tapes descussatus L. 6 pp. In FAO Technical Conference on Marine Pollution and Its Effects on Living Resources and Fishing. Rome, Italy, 9-18 December.

The accumulation of mercury (as $^{203}\text{Hg Cl}_2$) in Tapes descussatus (=japonica) in seawater was tested in two separate experiments. For both experiments, a rapid initial uptake was observed, raising the mercury concentration about ten times ambient levels within one day. After one day, accumulation slowed down. Gills and visceral organs concentrated the most mercury, and very little was accumulated by adsorption onto organ surfaces. The loss of mercury after accumulation from phytoplankton and seawater showed a continuous loss lasting longer than two months. It was much more strongly retained when introduced by injection into the foot. Biological half-times were also determined.

80. Window, H.L.

1973. Mercury distribution in estuarine - nearshore environment. J. Wat. Ways Harb. Coast. Engin. Div. 99:257-265.

Estuarine and nearshore water samples were taken in the area of southeastern United States between Cape Romain, South Carolina and Jacksonville, Florida. Samples of salt marsh grass (Spartina alterniflora) and sediment were taken from salt marshes along the Coast of Georgia, and plankton samples were taken off the northeastern and southeastern coast from surface waters nearshore and offshore. Mercury concentrations were measured by flameless atomic absorption. In nearshore waters mercury concentrations were low (0.01 - 0.1 ppb) and showed no significant variations between summer and winter. Higher values were observed near river discharges. Estuarine waters varied over wider limits and reached higher concentrations. Sediments from the salt marshes were relatively uniform and low with higher values occurring in nearby channel sediment where they probably reflect the mercury concentrations in the suspended sediments. Spartina appeared to concentrate mercury above the amount in sediments, especially in the roots. Plankton from nearshore polluted environments had much higher concentrations of mercury than offshore samples. They concluded that most of the mercury enters the estuarine system as a solution. About 50% of that which comes in as suspended particulates is deposited in salt marsh sediments. Plants take up significant amounts of mercury and appear to be an important means of biological transport to other compartments of the system. Mercury is transported to offshore areas only where river discharges are great enough to carry it through the system rapidly.

81. Windom, H.L. and R.G. Smith.

1972. Distribution of iron, magnesium, copper, zinc, and silver in oysters along the Georgia coast. J. Fish. Res. Bd. Canada 29:450-451.

The shells and soft tissues of approximately 50 oysters (Crassostrea virginica Gmelin) collected along the Georgia coast were analyzed for iron, magnesium, copper, zinc, and silver. Magnesium was the only metal found to be more concentrated in the shells. Correlations between the metals in the soft tissues suggest that iron and magnesium, and copper, zinc, and silver form two separate groups of metals which are concentrated similarly by the oyster. Geochemical characteristics of a given metal are an important control in their uptake by the oyster.

82. Young, D.R. and T.R. Folsom.

1967. Loss of Zn^{65} from the California sea-mussel Mytilus Californianus. Biol. Bull. 133:438-447.

To determine the loss rate of Zn^{65} in the California sea-mussel, colonies of Mytilus californianus averaging 11 cm. in length were translocated from the vicinity of the Columbia River mouth to southern California waters, where much lower levels of Zn^{65} occur. The colonies were suspended in nets at Scripps Institution of Oceanography Pier, and were sampled periodically for one year. During this time, the concentration of Zn^{65} in the soft tissues (after deducting radioactive decay) decreased by more than 97%. This decrease appears to be described satisfactorily by a single exponential function with a biological half-time for growing organisms of 76 ± 3.5 days. If radioactive decay is not excluded, the observed loss rate of Zn^{65} is described by an "effective half-life" of 58 ± 2.7 days.

83. Young, D.R.

1971. Mercury in the environment: a summary of information pertinent to the distribution of mercury in the Southern California Bight. Southern California Coastal Water Research Project (SCCWRP). 31 pp.

This is a review of mercury as an environmental pollutant with emphasis on the Southern California Bight including inventory and inputs of mercury, chemistry of mercury, mercury and public health, its effects on the health of marine organisms, and research recommendations.

C. SYNTHETIC ORGANICS AND PETROLEUM HYDROCARBONS

84. Albone, E.S., G. Eglinton, N.C. Evans, J.M. Hunter, and M.M. Rhead.
1972. Fate of DDT in Severn Estuary sediments. Environ. Sci. Tech. 6:914-918.

The functions of estuarine sediments as pollutant sink and as pollutant bank are assessed in relation to the fate of DDT in the environment. *p,p'*-DDT was degraded more slowly when incorporated in situ in Severn estuary sediments than when incubated in sediment samples maintained under hydrogen in the laboratory. These transformations are compared with the more extensive degradation of DDT on incubation in anaerobic sewage sludge. In all incubations, metabolites included *p,p'*-DDT. The wider application of the techniques developed is discussed.

85. Burnett, R.
1971. DDT residues: distribution and concentrations in Emerita analoga (Stimpson) along coastal California. Science 174:606-608.

The total concentrations (tDDT) of DDT (1,1,1-trichloro-2,2-bis(p-chlorophenyl)ethane), DDD (1,1-dichloro-2,2-bis(p-chlorophenyl)ethane), and DDE (1,1-dichloro-2,2-bis(p-chlorophenyl)ethylene) in Emerita analoga from 19 California beaches reflect tDDT contamination nearby. Animals near the Los Angeles County sewer outfall contain over 45 times as much tDDT as animals near major agricultural drainage areas. Sediments near the outfall probably contain over 100 metric tons of tDDT--a reservoir for input into marine organisms. The effluent from a plant that manufactures DDT is a probable source.

86. Butler, P.A.
1973. Organochlorine residues in estuarine mollusks, 1965-72--national monitoring program. Pest. Monit. J. 6:238.

This paper describes the development of the national program for monitoring estuarine mollusks in 15 coastal States and reports the findings for the period 1967-72. The report is presented in two parts: Part I. General Summary and Conclusions, and Part II. Residue Data--Individual States.

Analyses of the 8,095 samples for 15 persistent organochlorine compounds showed that DDT residues were ubiquitous; the maximum DDT residue detected was 5.39 ppm. Dieldrin was the second most commonly detected compound with a maximum residue of 0.23 ppm. Endrin, mirex, toxaphene, and polychlorinated biphenyls were found only occasionally. Results indicate a clearly defined trend towards decreased levels of DDT residues, beginning in 1969-70. At no time were residues observed of such a magnitude as to imply damage to mollusks; however, residues were large enough to pose a threat to other elements of the biota through the processes of recycling and magnification.

87. Cox, J.L.

1971. DDT residues in seawater and particulate matter in the California current system. Fish. Bull. 69:443-450.

Continuous samples of seawater and organic particulate material collected along linear transects in the California current system were analyzed for DDT residues. DDT residue concentrations in whole seawater, as determined by continuous-flow, liquid-liquid extraction, ranged from 2.3×10^{-12} g/ml off Oregon and Washington, to 5.6×10^{-12} g/ml off southern California. Geographical patterns in these concentration values are discussed in relation to mechanisms of land-sea DDT residue transfer. DDT residue concentrations in particulate material collected by continuous-flow centrifugation and filtration of the centrifugal pellet onto GFC-glass-fiber filters ranged from 1.2 to 5.7×10^{-6} g/g carbon (with one exception). These values were related to the density of the standing crops. DDT residues in this particulate fraction accounted for less than 10% of the DDT residues in the whole seawater samples. Residues which are fixed to particles of less than 1-2 μ in diameter may account for the balance of the DDT residues in the whole water samples. Experimental results are described which implicate adsorption as the uptake mechanism for algal cells; these experiments also support the idea that less than 1-2 μ diameter particles carry most of the DDT residues in whole seawater.

88. Croker, R.A. and A.J. Wilson.

1965. Kinetics and effects of DDT in a tidal marsh ditch. Trans. Amer. Fish. Soc. 94:152-159.

A tidal marsh ditch near Pensacola, FL was treated with 0.2 pound of DDT per acre in March 1963, and observed for 4 months. Ninety-eight per cent of the animal mortality occurred within 3 weeks after treatment. A population of cyprinodont fish was restored by reproduction of surviving and introduced fish. DDT was detected in surface water samples 8 days after treatment but not after 14 days, while DDT was not detected in bottom water samples

later than 1 day after treatment. Seven weeks after treatment, vegetation samples averaged 9.1 ppm of DDT, after reaching a maximum of 75 ppm of DDT 3 to 4 weeks after treatment. Sediment samples yielded more variable residues, averaging 0.76 ppm of DDT at 7 weeks after reaching a maximum of 3.35 ppm of DDT 6 weeks after treatment. Fish accumulated up to 90 ppm of DDT within 5 weeks after treatment. Highest DDT residues detected in fiddler crabs and snails ranged from 15 to 25 per cent of the average residues detected in fish. The net water movement as indicated by the distribution of Rhodamine B dye, the general distribution of DDT residues in all samples, and the difference in mortality of fish at two holding sites, indicated an accumulation of DDT in the lower marsh after treatment.

89. DiSalvo, L.H. and H.E. Guard.

1975. Hydrocarbons associated with suspended particulate matter in San Francisco Bay waters. Presented at Conference on Prevention and Control of Oil Pollution. March 25-27. San Francisco, CA.

Suspended sediments were obtained at seven stations in San Francisco Bay during the summer of 1974 using a double settling tube device termed the "biosampler". One tube of the device passively collected suspended sediments which settled from ambient waters at the sampling sites. The top of the second tube contained bay mussels (Mytilus edulis) as biological agents for the active entrapment and deposition of suspended particulates occurring in the water. Presence of the mussels in the sampler was, in most cases, indispensable for collection of sufficient amounts of material for analysis over one-week sampling periods.

A thin-layer chromatographic method was employed for analysis of total alkane and total aromatic hydrocarbons in recovered sediments. The sediments were found to contain 190-6188 ppm dry weight of total hydrocarbons, with alkane-aromatic ratios varying from 1.1 to 5.1. Water separated from recovered sediments after shaking contained from 15 to 450 µg per liter total hydrocarbons. Filtration of these water samples through 0.45 Millipore filters had little or no effect on their hydrocarbon concentration.

Calculations based on minimum possible values suggested that 13.5 or more metric tons of presumably pollutant hydrocarbons were present in association with suspended particulate matter in the bay system at any given time during the sampling period.

Previously published information on bay circulation suggested that suspended particulates, and thus pollutant hydrocarbons, may be accumulated in the shoal areas of the eastern bay margins.

90. Disalvo, L.H., H.E. Guard, and L. Hunter.

1974. Tissue hydrocarbon burden of mussels as a potential monitor of environmental hydrocarbon insult. Annual Progress Report for 1973. ONR Contract NO0014-69-A-0200-1001. 19pp.

A survey method for the analysis of total tissue hydrocarbon burden has been used to measure hydrocarbon concentrations in the mussels Mytilus edulis and Mytilus californianus. Organisms transplanted from cleanwater stations to polluted water stations took up hydrocarbons and when they were replaced in clean waters, their hydrocarbon content approached cleanwater baselines. Organisms transplanted from polluted water to clean water lost a minor fraction of their hydrocarbon burden over a 10-week period and suggested a return to baseline values when replaced in polluted waters of origin. The mussels and perhaps other invertebrate animals appear to be useful as systems monitors of chronic hydrocarbon pollution on a relative basis.

91. Goerlitz, D.F. and L.M. Law.

1974. Distribution of chlorinated hydrocarbons in stream-bottom material. J. Res. U. S. Geol. Surv. 2:541-543.

The distribution of some chlorinated hydrocarbons was studied in bed material samples gathered from selected surface streams which had been contaminated with high levels of the compounds investigated. Separate fractions of each sample were analyzed. Analysis of the water used for fractionation showed a loss of less than 1% of the chlorinated hydrocarbon to the aqueous phase. The expectation that the greatest concentration of chlorinated hydrocarbons in the various fractions would be related to the surface area, i.e., on the finer material, was not always correct. The greatest amounts of the subject compounds were found in the coarser fractions of 4 of the samples. Although surface area generally related to sorption, notable exceptions occurred. The complexity of bottom material samples is such that the distribution of chlorinated hydrocarbons may be controlled to a major extent by the organic matter and associated organism. Physical and mineralogical considerations may be of less importance. This investigation shows that all particle-size fractions must be considered if a representative bed-material is to be collected and quantitatively related to a stream.

92. Halcrow, W., D.W. MacKay, and J. Bogan.

1974. PCB levels in Clyde marine sediments and fauna. Mar. Poll. Bull. 5:134-136.

The disposal of sewage sludge-containing PCBs in the Firth of Clyde has resulted in heavy contamination of the sediments in a confined area. The distribution of PCBs in the disposal area parallels that of a number of other pollutants. There is also a more widespread low-level contamination of surface sediments, probably as a result of PCBs being transported in association with fine organic particulate material. The contamination of the benthic fauna reflects that the levels of PCB in the sediments may contribute to the reduced faunal diversity in the

center of the disposal area. With the restrictions imposed by the manufacturers on the use of PCBs, and the elimination of other sources in the Clyde catchment area, the levels of PCBs in the Garroch Head area are expected to diminish. A 66% reduction in levels present in the sewage sludges has already been reported. Monitoring of PCB levels in the Garroch Head area will be continued over the next few years.

93. Hamelink, J.L., R.C. Waybrant, and R.C. Ball.

1971. A proposal: exchange equilibria control the degree chlorinated hydrocarbons are biologically magnified in lentic environments. Trans. Amer. Fish. Soc. 100:207.

The dynamics of DDT in lentic environments were studied in a farm pond and four artificial pools. A hypothesis that biological magnification of pesticides was dependent on passage of residues through a food chain was rejected and a hypothesis that accumulation depends on adsorption and solubility differences was proposed. The mechanism proposed is based on the principle that compounds are exchanged between water and fats. Exchange in fish passes through two stages, from water to blood and from blood to fats, permitting a high degree of magnification in fish. The proposed mechanism accounts for the reported observations that pesticides are excreted by fish, that body load of pesticides increases as the fat content of fish increases, that pesticide magnification by fish is inverse to the water solubility of the compounds and pesticides persist longer in oligotrophic than eutrophic lentic ecosystems.

94. Huang, J.C.

1971. Effect of selected factors on pesticide sorption and desorption in the aquatic system. Wat. Poll. Contr. Fed. J. 43:1739-1748.

Three pesticides (dieldrin, heptachlor and DDT) were selected to investigate sorption and desorption of chlorinated hydrocarbons by clay minerals (montmorillonite and illite). The effects of pH, temperature, salinity, and organic pollutants on the adsorption rate and equilibrium were the major considerations.

From the results of this study the following conclusions may be drawn: (1) At the test pH range of 6.0 to 10.0, the adsorption of dieldrin by montmorillonite was slightly increased with a decreasing pH. (2) At the normal temperature range of 10° to 30°C the adsorption and desorption of dieldrin by the montmorillonite sediment was not significantly affected by changes in water temperature. (3) The effect of the salt (NaCl) concentration of water on the dieldrin adsorption and desorption by montmorillonite

was not conclusive. Slightly more pesticide was adsorbed and retained more strongly by the clay at a salt content of 3 percent than at 0.3 or 0.03 percent; however, the same pesticide was adsorbed slightly more and retained more strongly at a salt content of 0.03 than at 0.3 percent. (4) Several representative organic pollutants such as glucose, alanine, and stearic acid were found to exert no effect at all on the rates and equilibria of the adsorptions of dieldrin, heptachlor, and DDT by montmorillonite and illite. (5) The dieldrin adsorption by montmorillonite was not influenced by the soluble organic matter contained in filtered domestic wastewater.

95. Koeman, J.H., M.C. Ten Noever de Brauw, and R.H. de Vos.
1969. Chlorinated biphenyls in fish, mussels, and birds from the River Rhine and Netherland coastal area. *Nature* 221:1126-1128.

Polychlorinated biphenyls are widely used in industry, and they are now making an appearance in the world's wildlife. Besides being a hazard in their own right, they interfere with the detection of organochlorine insecticides such as DDT. Occurrence and distribution of PCBs in the River Rhine and the Netherlands coast are described. Results indicate that the lower chlorinated PCBs are less persistent.

96. Law, L.M. and D.F. Goerlitz.
1974. Selected chlorinated hydrocarbons in bottom material from streams tributary to San Francisco Bay. *Pest. Mont. J.* 8:33.

As part of a study of the environmental quality of San Francisco Bay, bottom material from 26 streams tributary to the Bay were analyzed for chlordane, DDD, DDE, DDT and PCB residues. These compounds were present in essentially all streams tested. Chlordane proved to be ubiquitous, with a concentration range similar to that of the other compounds. Noteworthy was the occurrence in one stream of polychlorinated naphthalene residues. Compounds occurring in concentration above 10 ug/kg were identified in most instances by combined gas chromatography/mass spectrometry.

97. Lee, R.F., R. Sauerheber, and A.A. Benson.
1970. Petroleum hydrocarbons: uptake and discharge by the marine mussel, Mytilus edulis. *Science* 177:344-346.

The common marine mussel Mytilus edulis has been observed to rapidly take up mineral oil, (¹⁴C)heptadecane, 1,2,3,4-tetrahydronaphthalene, (¹⁴C)toluene, (¹⁴C)naphthalene, and (³H)3,4-benzo-

pyrene from seawater solution. This species of mussel did not metabolize any of these compounds and transfer of the mussel to fresh seawater, after exposure to the hydrocarbon in solution, resulted in the discharge of most of the hydrocarbon, although significant amounts remained (between 1 and 400 micrograms per mussel). The nontoxic paraffinic hydrocarbons mineral oil and heptadecane were taken up (10 milligrams per mussel) to a much greater extent than the aromatic hydrocarbons (2 to 20 micrograms per mussel).

98. Leland, H.V., W.N. Bruce, and N.F. Shimp.

1973. Chlorinated hydrocarbon insecticides in sediments of southern Lake Michigan. *Environ. Sci. Tech.* 7:833-838.

Past usage of organochlorine insecticides in states bordering southern Lake Michigan has resulted in the presence in the lake sediments of large reservoirs of DDT, DDE, DDD and dieldrin. Analysis of surficial sediments from different areas of southern Lake Michigan indicated high positive statistical correlations between tDDT, (DDT, DDE, DDD) and dieldrin concentrations in the sample and organic carbon content. The amount of organic carbon in surficial sediments was directly proportional to the clay-size fraction of sediment. Dechlorination of DDT to form DDD apparently predominates under the reducing conditions of sediments on the eastern side of the south basin of Lake Michigan. In other areas, DDT is the principal component of the tDDT complex in sediments and DDE and DDD are higher degradative products.

99. Leshniowsky, W.O., R.R. Dugan, R.M. Pfister, J.I. Frea, and C.I. Randles.

1970. Adsorption of chlorinated hydrocarbon pesticides by microbial floc and lake sediment and its ecological implications. *Proc. 12th Conf. Great Lakes Res.* 611-618.

Of thirty-eight aerobic bacteria isolated from Lake Erie, 14 formed flocs in at least one of six different media used. Two of these floc formers were examined for ability to accumulate aldrin from solution. Aldrin (10^{-6} g/ml) was dissolved in acetone and added to flasks containing pregrown bacterial flocs suspended in water. Flocs were shaken for various time intervals and separated from solution by centrifugation. Both were analyzed separately for presence of aldrin, using gas liquid chromatography.

Contemporary sediment collected from Lake Erie was examined microscopically, analyzed for pesticide content and ability to adsorb aldrin.

Bacterial flocs adsorbed aldrin from solution giving a 625X concentration factor within 20 min. after which there was no further increase. The collected sediment behaved similarly.

Floc forming microbes settling from a water column remove pesticides and represent a natural purification process. The

pesticides may then accumulate in bottom sediments and exert a toxic effect on susceptible fauna.

100. MacKay, D. and A.W. Wolkoff.

1973. Rate of evaporation of low-solubility contaminants from water bodies to atmosphere. Environ. Sci. Tech. 7:611.

Equations are derived to predict the rate of evaporation from aqueous solutions of compounds such as hydrocarbons and chlorinated hydrocarbons which are of low solubility. The rate of evaporation can be high even for compounds of low vapor pressure and "half lives" in solution can be as low as minutes or hours under laboratory or environmental conditions. The rate may be limited by diffusion or desorption. Transfer of contaminants from the water to air environments may thus occur much faster than has been generally appreciated.

101. Metcalf, R.L., J.R. Sanborn, P.Y. Lu, and D. Nye.

1975. Laboratory model ecosystem studies of the degradation and fate of radiolabeled tri-, tetra-, and pentachlorobiphenyl compared with DDE. Arch. Environ. Contam. Toxic. 3:151.

Radiolabeled tri-, tetra-, and pentachlorobiphenyls (PCB) and DDE were studied in a laboratory model ecosystem for degradation pathways, and biomagnification in alga, snail, mosquito, and fish. Trichlorobiphenyl was degraded in all the organisms of the model ecosystem much more rapidly than tetrachloro- and pentachlorobiphenyl. Pentachlorobiphenyl was approximately as persistent as DDE. There was a linear relationship between lipid/water partition and ecological magnification and between water solubility and ecological magnification. No evidence of conversion of DDE to PCB was detected.

102. Ninuno, D.R., R.R. Blackman, A.J. Wilson, Jr. and J. Forester.

1971. Toxicity and distribution of Aroclor 1254 in the pink shrimp Penaeus duorarum. Mar. Biol. 11:191-197.

The polychlorinated biphenyl Aroclor 1254 was released in an accidental leakage of heat-exchange fluid from an industrial plant, into the Escambia River, near Pensacola, Florida, USA. This material was carried downstream, and is now found in the fauna of Escambia Bay and its contiguous waters, prime nursery areas for fishes and invertebrates such as penaeid shrimp. The significance of pollution by this chemical was assessed by establishing toxicity levels, determining routes of entry, and investigating its movement and distribution in various tissues of shrimp

under controlled conditions in the laboratory. Aroclor 1254 added to the water was toxic to the juvenile pink shrimp Penaeus duorarum at a concentration of 1.0 part per billion within 15 days, but was less toxic to adult pink shrimp. Shrimp obtained the contaminant from water and food and concentrated it to 510.0 parts per million in the hepatopancreas. Aroclor 1254 residue data from shrimp collected in the estuary are included in the study.

103. Nimmo, D.R., P.D. Wilson, E.R. Blackman, and A.J. Wilson, Jr.
1971. Polychlorinated biphenyls absorbed from sediments by fiddler crabs and pink shrimp. *Nature* 231:50-52.

Fiddler crabs and shrimp exposed to the contaminated sediments accumulated Aroclor 1254 in their tissues by ingesting contaminated sediment particles or by absorbing the leached chemical from water. They sifted the sediments before ingesting the particles. In most cases, the amount of Aroclor in individual crabs or in shrimp hepatopancreas was directly related to the amount in the sediments. We analysed the distribution of the chemical in the sediment in relation to particle size and found it was evenly distributed. Large concentrations of Aroclor residues may also accumulate in the hepatopancreas of shrimp exposed to sandy silts because the chemical leaches from the sediments and is absorbed through the gills. Three and a half part per billion (ppb) were in the effluent water from the aquarium with sandy silt (61.0 ppm Aroclor); the water from the silt (30.0 ppm) had only 0.5 ppb. Even though fiddler crabs were not continuously covered with water, they could have obtained Aroclor from the water as they wetted their gills. We have demonstrated experimentally that Aroclor 1254 can enter the estuarine food chain from sediments.

104. Nimmo, D.R., D.J. Hansen, J.A. Couch, N.R. Cooley, P.R. Parrish, and J.I. Lowe.
1975. Toxicity of Aroclor 1254 and its physiological activity in several estuarine organisms. *Arch. Environ. Contam. Toxic.* 3:22.

The occurrence of high concentrations of PCB (Aroclor 1254) in the Pensacola estuary prompted field and laboratory studies by the Gulf Breeze Environmental Research Laboratory (EPA). Monitoring of the estuary indicates the chemical is present in all components--particularly in sediments and fishes. Residues appear to be diminishing in sediments. Toxicity tests show estuarine species sensitive at ppb concentrations in water, with a ciliate protozoan (Tetrahymena pyriformis W), shrimps (Penaeus duorarum, P. aztecus, and Palaemonetes pugio), and a fish (Fundulus similis) affected at or near 1.0 ppb. Tissue concentrations of Aroclor 1254 similar to those found in natural populations of shrimps from the contaminated estuary were successfully duplicated in laboratory experiments. Shrimps also concentrated the PCB from very low

concentrations (0.04 ppb) in the water. Three estuarine species demonstrated pathologic changes at tissue and cellular level after chronic exposure to the chemical. Oysters (Crassostrea virginica) developed abnormal infiltration of leukocytes in the connective tissue, spot (Leiostomus xanthurus) developed fatty changes in their livers, and shrimp (Penaeus duorarum) developed crystalloids in hepatopancreatic nuclei.

105. Oloffs, P.C., L.J. Albright, and S.Y. Szeto.
1972. Fate and behavior of five chlorinated hydrocarbons in three natural waters. Canadian J. Microbiol. 18:1393-1398.

The data from this study show that surface-active pollutants in natural waters, e.g. industrial and domestic detergents, may significantly influence the behavior of highly insoluble chlorinated hydrocarbon residues.

The means by which chlorinated hydrocarbon residues are moved from points of release to the open ocean are still not completely understood, but data from this investigation suggest some possibilities: (1) When they are present in natural waters at concentrations approaching or exceeding their solubilities, they appear to accumulate at the water-air interface from whence they evaporate into the atmosphere. Under these conditions, the rates of evaporation exceed those explainable by codistillation. (2) If these compounds occur at concentrations which are lower than their solubilities, they are evenly distributed and their rate of transfer from the water will be lower, about proportional to the evaporation of water, i.e. they transfer by codistillation. (3) The presence of materials which promote the even distribution of chlorinated hydrocarbon residues in natural waters will decrease the transfer of residues into the atmosphere because they counteract accumulation at, and thus evaporation from, the water-air interface.

106. Pierce, R.H., Jr., C.E. Olney, and G.T. Felbeck, Jr.
1974. p,p'-DDT adsorption to suspended particulate matter in seawater. Geochim. Cosmochim. Acta 38:1061-1073.

A study was made to determine what fraction of suspended particulate matter is responsible for adsorbing p,p'-DDT from sea water and to investigate the mechanism of adsorption to that fraction.

For the model system used here, an equilibrium concentration of 0.1 ppb DDT in distilled water and a suspended load of 10 mg/l. consisting of 80% montmorillonite clay and 20% humic acid was assumed to approximate a polluted estuarine system. In 1 liter of this system, the amount of DDT adsorbed to montmorillonite clay would be 34 ng (4200 ppb) and to humic acid, 152 ng (76,000 ppb). Therefore about 35% of the total DDT would be in solution, 12% would be adsorbed to clay and 53% to humic acid. This study supports the theory that DDT adsorption occurs by electrostatic attraction between hydrogen atoms on the aromatic rings and negatively charged sites on the clay surface to DDT molecules ranging from easily accessible surface sites to less accessible internal voids. In a polar solvent, as water, the affinity of DDT for hydrophobic surfaces is enhanced. It is concluded that the suspended humic matter is the agent most responsible for scavenging chlorinated hydrocarbons from sea water and transporting them through the water column.

107. Vreeland, V.

1974. Uptake of chlorobiphenyls by oysters. Environ. Poll. 6:135-140.

Accumulation of individual chlorinated biphenyls in small oysters was proportional to the degree of chlorination of the isomer and to its concentration in seawater at environmental levels. Equilibrium was reached after one month, with concentration factors of 1200 to 48,00 for isomers with 2 to 6 chlorine atoms per molecule. Partitioning of PCB between hydrophobic lipids and ambient seawater has been rejected as the uptake mechanism.

108. Woodwell, G.M., C.F. Wurster, Jr., and P.A. Isaacson.

1967. DDT residues in an East Coast estuary: a case of biological concentration of a persistent insecticide. Science 156:821-3.

DDT residues in the soil of an extensive salt marsh on the south shore of Long Island averaged more than 13 pounds per acre (15 kilograms per hectare); the maximum was 32 pounds per acre (36 kilograms per hectare). A systematic sampling of various organisms from the vicinity showed concentrations of DDT increasing with trophic level through more than three orders of magnitude from 0.04 part per million in plankton to 75 parts per million in a ring-billed gull. Highest concentrations occurred in scavenging and carnivorous fish and birds, although birds had 10 to 100 times more than fish. These concentrations approach those in animals dying from DDT poisoning, which suggests that many natural populations in this area are now being affected, possibly limited, by DDT residues. Similar concentrations have been reported elsewhere in North America.

109. Young, D.R. and T.C. Heesen.

1974. Inputs and distributions of chlorinated hydrocarbons in three southern California harbors. Southern California Coastal Water Research Project (SCCWRP). Report TM214.

Chlorinated hydrocarbons such as the pesticide DDT and industrial polychlorinated biphenyls (PCB) are major contaminants in southern California marine waters. As a result of a predominant DDT input from the large submarine discharge of Los Angeles County municipal wastewater, coastal mussels off Los Angeles contain up to 30 times more p,p'-DDE than do those off San Diego. Specimens of a bay mussel collected from San Diego Bay, Newport Harbor, and San Pedro Harbor also show this pattern of increasing DDT concentrations toward Los Angeles. However, PCB 1254 concentrations are similar in mussels from the three harbors and are several times higher than in specimens collected from the nearby coastal waters. Estimates for total annual inputs of PCB 1254 to these harbors from municipal wastewater, industrial wastewater, surface runoff, aerial fallout, and vessel antifouling paints range from 1 kg/yr in Newport Bay to 150 kg/yr in San Pedro Harbor, with surface runoff and industrial wastewater constituting virtually all of the latter input. Although antifouling paints presently constitute a completely insignificant mode for PCB input to these harbors, occasional high PCB concentrations in old paint chips and the correlation of mussel PCB concentrations with antifouling paint usage suggest that this may have been the predominant source of PCB to southern California harbors in recent years. It is not yet known how long these harbors will exhibit PCB contamination levels higher than those of the adjacent coastal waters.

110. Young, D.R., D.J. McDermott, T.C. Heesen, and T.K. Jan.

1975. Pollution inputs and distributions off southern California. Reprint of paper presented at 169th National Meeting of ACS in Special Symposium, Marine Chemistry in the coastal environment, 8-10 April. Philadelphia, PA.

Municipal wastewaters are the dominant known source of numerous trace contaminants to the Southern California Bight. Annual inputs of mercury, DDT, and PCB via this route are an order of magnitude larger than those from direct industrial discharges and storm runoff. Aerial fallout is also a major source of chlorinated hydrocarbons. Bottom sediments around the largest outfall system are highly contaminated by various trace elements and DDT residues. Corresponding contamination of benthic organisms occurs for DDT but not the trace elements studied. Seawater collected from the mouths of three important harbors contained copper and the chlorinated hydrocarbons at the part-per-billion and part-per-trillion level, respectively.

D. SEDIMENTS AND WATER

111. Brooks, R.R., B.J. Presley, and I.R. Kaplan.

1968. Trace elements in the interstitial waters of marine sediments. *Geochim. Cosmochim. Acta* 32:397-414.

Phosphate and eleven elements were determined in the interstitial waters of four piston cores from the continental borderland area off the coast of Southern California. Calcium, magnesium, potassium, sodium and strontium were determined directly by atomic absorption spectrophotometry. Phosphate was determined colorimetrically and the trace elements: cadmium, cobalt, copper, iron, nickel and zinc were determined by extraction of the chelate complexes with ammonium pyrrolidine dithiocarbamate into methylisobutyl ketone and subsequent analysis by atomic absorption spectrophotometry. Correlation coefficients for various pairs of variables are given.

Decrease of Eh and increase of phosphate with depth was attributed mainly to bacterial activity and, in the case of phosphate, partly due to release from metal phosphates.

Calcium and strontium showed decreases with depth probably due partly to precipitation of carbonate by excess bicarbonate ions produced as a result of bacterial activity and partly to precipitation as phosphate.

Potassium levels were higher and magnesium values were slightly lower than in sea water, probably due to interaction with the solid phase of the sediments.

Copper showed no overall trends, but zinc showed moderate decrease with the depth and correlated well with cadmium. Enrichment of cobalt, zinc, cadmium and copper at the surface of the sediments was attributed to biological concentration and release, since there is more than sufficient organic matter present to account for such a source.

Iron and nickel concentrations were greater in reducing sediments and showed an overall trend to increase with depth.

112. Carmody, D.J., J.B. Pearce and W.E. Yasso.

1973. Trace metals in sediments of New York Bight. *Mar. Poll. Bull.* 4:132-136.

Concentrations of chromium, copper, lead, nickel and zinc in superficial sediments in New York Bight are ten to a hundred times greater near waste disposal areas than in uncontaminated sediments. Some wastes are transported by currents north-east towards Long Island and south-east along the Hudson Submarine Valley.

113. Chow, T.J., K.W. Bruland, K. Bertine, A. Soutar, M. Koide, and E.D. Goldberg.
1973. Lead pollution: records in Southern California coastal sediments. *Science* 181:551-552.

The present anthropogenic lead fluxes into sediments from Santa Monica, San Pedro, and Santa Barbara basins of Southern California are, respectively, 0.9, 1.7, and 2.1 micrograms of lead per square centimeter of sea bottom per year; the natural (prepollution) rates for these three basins were, respectively, 0.24, 0.26, and 1.0 microgram of lead per square centimeter per year. Studies of isotopic composition indicate that lead pollutants in coastal sediments are derived mainly from the combustion of lead additives in gasoline.

114. Cranston, R.E. and D.E. Buckley.
1972. Mercury pathways in a river and estuary. *Environ. Sci. Tech.* 6:274-278.

A method for measuring total mercury in water, suspended particulate matter, and bottom sediments has been evaluated. Some data have been applied to a study of the geochemical pathways of mercury in a rural river and estuary system. Concentrations of mercury in the LaHave River (Nova Scotia) are related to the proximity of a small rural town. The dissipation of mercury in solution appears to be through dilution, as well as by adsorption on suspended particulate matter which raises the level in particulate matter to the range of 2.04-34.4 ppm. Bottom sediments in the LaHave River are affected by sedimentation of particulate matter containing high levels of mercury, but the mercury concentration in the bottom sediments ranges from 0.09 to 1.06 ppm. Mercury released to the natural environment from industrial waste effluents appears to be discharged mostly in the dissolved form but may be quite rapidly adsorbed as shown by analyses of suspended particulate matter and bottom sediments.

115. DeGroot, A.J., J.J.M. DeGoeij, and C. Zegers.
1971. Contents and behavior of mercury as compared with other heavy metals in sediments from the Rivers Rhine and Ems. *Geologie en Mijnbouw* 50:393-398.

Mercury is one of the various waste substances transported by the river Rhine across the German-Dutch border. Together with a number of other heavy metals, mercury is present in large quantities. Upstream these metals are predominantly fixed to the suspended solids in the water, and may be deposited on river flats and flood plains. From the fresh-water tidal area of the river onward, however, these elements are more or less solubilized during their transport as organo-metallic complexes. The mobilization of mercury is most pronounced in this respect, leading to more normal amounts of this element in sediments from the Wadden Sea. This article deals with the behaviour

of mercury through the whole Rhine estuary as compared with a number of other heavy metals.

As a counterpart of the Rhine the same processes are described for the river Ems. The latter may be regarded as a classic example of an unpolluted stream.

116. Duce, R.A., J.G. Quinn, C.E. Olney, S.R. Piotrowicz, B.J. Ray and T.L. Wade.

1972. Enrichment of heavy metals and organic compounds in the surface microlayer of Narragansett Bay, Rhode Island. *Science* 176:161-163.

Concentrations of lead, iron, nickel, copper, fatty acids, hydrocarbons, and chlorinated hydrocarbons are enriched from 1.5 to 50 times in the top 100 to 150 micrometers of Narragansett Bay water relative to the bulk water 20 centimeters below the surface. Trace metal enrichment was observed in the particulate and organic fractions but not in the inorganic fractions. If these substances are concentrated in films only a few molecular layers thick on the water surface, the actual enrichment factor in the films may be well over 10^4 resulting in extremely high localized pollutant concentrations in the surface microlayer.

117. Duchart, P., S.E. Calvert, and N.B. Price.

1973. Distribution of trace metals in the pore waters of shallow water marine sediment. *Limnol. Oceanogr.* 18:605-10.

The concentrations of seven trace metals in the pore waters of marine sediments from Loch Fyne, Scotland, have been analyzed by atomic absorption spectrophotometry following solvent extraction. The samples were taken from sediments containing variable amounts of sulfide at depth and variable thicknesses of oxides at the surfaces; one core contains abundant manganese nodules and manganese-calcium carbonate concretions.

The distribution of dissolved Mn shows a regular downward decrease in all cores examined which is interpreted as an indication of recycling of Mn between solid diagenetic mineral phases and pore solutions. Dissolved Fe, in lower concentrations, either decreases or increases down the core depending on the type of sediment; Cu, Ni, Zn, Co, and Pb also show variable concentration profiles indicating control by different postdepositional reactions in the sediment, which in turn depend on different physiochemical environments generated by variable amounts of contained organic material and different accumulation rates.

118. Du Pleissi , S.F. and R. du T. Burger.

1971. The displaceability of specifically adsorbed copper from clay minerals. *Agrochemophysica* 3:17-22.

The displaceability of specifically adsorbed copper from kaolinite and montmorillonite is determined by the size of the displacing ion. Ions with approximately the same size as the Cu-ion eg. Co^{++} were the most successful displacers of Cu^{++} . Cu^{++} is presumably partly adsorbed in the clay structure of the minerals viz. the Si-tetrahedral layer. Due to the natural rotation of the tetrahedra, the size of the holes can be reduced up to one third, which will favour the possibility of adsorption of copper in this position.

A part of the copper is probably also adsorbed due to exchange with weakly dissociated H-ions, thereby explaining the reduction in pH which occurred in some cases when copper was adsorbed.

119. Gardiner, J.

1974. The chemistry of cadmium in natural water--II. The adsorption of cadmium on river muds and naturally occurring solids. *Wat. Res.* 8:157-164.

The adsorption of cadmium on mud solids and particles of clay, silica, humic material and other naturally occurring solids has been studied. Radiochemical methods were employed so that cadmium concentrations in the $\mu\text{g l}^{-1}$ range could be used. The variation of the extent of adsorption with many of the large number of factors involved was investigated, concentration factors (distribution coefficients) were determined, and in the course of the work, loss of cadmium by adsorption on container surfaces and filters was investigated.

Rates of adsorption and desorption were shown to be rapid. Concentration factors for river muds varied between 5000 and 50,000 and depended mainly on the type of solid, its state of subdivision, the concentration of metal ion present, the time of contact and the concentration of complexing ligands. Humic material appeared to be the main component of river mud responsible for adsorption. Adsorption and desorption processes are likely to be major factors in controlling the concentration of cadmium in natural waters and will tend to counteract changes in the concentration of the metal ion in solution.

120. Goldberg, E.D.

1954. Marine Geochemistry. I--Chemical scavengers of the sea. *J. Geol.* 62:249-265.

The ability of the hydrated oxides of manganese and iron to adsorb ions from solution (scavenging) is considered in relation to

some problems in marine geology, chemistry, and biology. In the ferruginous sediments of the Pacific Ocean, iron oxides are accompanied by titanium, cobalt, and zirconium in amounts proportional to the iron content. Similarly, copper and nickel are linearly related to the manganese content. These observations are explained on the basis of scavenging. An electrochemical theory for the formation of manganese nodules is presented. Marine sediments are classified on the basis of the geosphere in which the solid phases originate. The distribution of certain ionic species in sea water between the solid and aqueous phases is considered on the basis of scavenging and co-ordination compound theory. The concentration of minor elements by members of the marine biosphere is explained either by the direct uptake of the element or by the uptake of iron or manganese oxides with the accompanying scavenged element.

121. Gorham, E. and D.J. Swaine.

1965. The influence of oxidizing and reducing conditions upon the distribution of some elements in Lake Sediments. *Limnol. Oceanogr.* 10:268-279.

Analyses for Fe, Mn, Pb, Zn, Co, P, Mo, Ba, Sr, Ti, Li, Rb, Na, K, Be, Cr, V, Y, Ag, Cu, C, S, Sn, Ni, Ga, Zr, and La have been carried out on some oxidate crusts, oxidized surface muds, reduced subsurface muds, and glacial clays collected in Windermere and Esthwaite Water in the English Lake District. The relatively organic lake muds exhibit the highest concentrations of C, S, Cu, Sn, and Ni. Many of the oxidate crusts exhibit strong enrichment in Mn, Fe, Ba, Sr, Pb, and Zn. Ti, Li, Rb, Co, P, and Mo are also enriched in some crusts. S, Sn, and Ni reach their highest levels in the reducing subsurface muds, but Mn, and to a lesser extent Fe and Mo, are higher in the oxidized than in the reduced muds.

Relations between the elements in the various sediments are examined, and the English oxidate crusts are compared with lake and stream ores in Sweden and Finland, and with marine manganese nodules. These marine nodules are frequently enriched in Cu, Ni, Co, Mo, V, Ag, and Sn to a far higher degree than the freshwater ferromanganese concretions. Some freshwater crusts exhibit enrichments in Pb, Zn, and Ba of the same order as those observed in marine manganese nodules.

122. Gotoh, S. and W.H. Patrick, Jr.

1972. Transformation of manganese in a waterlogged soil as affected by redox potential and pH. *Soil Sci. Soc. Amer. Proc.* 36:738-742.

The distribution of different forms of manganese in waterlogged soil was studied over a wide range of closely controlled Eh-pH conditions. At pH 5 almost all of soil manganese was converted from the reducible to the water soluble plus exchangeable fraction

even at a redox potential as high as +500 mV. In sharp contrast, at pH levels between 6 and 8 most of the conversion took place at relatively lower redox potentials of +200 to +300 mV. When the water soluble plus exchangeable fraction was further divided into its two components, low pH, and low E_h were found to increase water soluble manganese at the expense of the exchangeable form. Cation exchange reactions were apparently important in regulating the equilibria between water soluble and exchangeable manganese. Labeled manganese added to the soil showed an almost identical distribution among the various manganese fractions to that of native soil manganese. The observed pMn/pH slope and E_o were much lower at pH 6 to 8 than those calculated for pure equilibrium systems. It may be concluded from this study that the E_h and pH of flooded soils provide general control of manganese transformation which probably include both chemical and biological processes, the complexity of which make it difficult to explain the reactions in terms of simple chemical systems.

123. Gustafson, J.F.

1972. Marine sediments, permanence of attachment of adsorbed metals and pesticides, San Leandro Bay, CA. Prepared for: Harbor Bay Isle Associates. 35 pp.

The major objective of this report was to present an experiment designed to explain the role of resuspension of sediments in availability of toxic materials to invertebrates. In the first experiment, one sample of Bay mud was resuspended in seawater with a resultant reduction in the concentration of heavy metals and pesticides in the water. This is explained by the scavenging effect of clays, which adsorb toxic materials and remove them to the bottom. The second experiment involved placing Mya arenaria in aquariums of control (clean) Bay sediment and resuspending the sediment with an electric motor and propeller, keeping the turbidity very high for 10 days. The results showed the clams in bay sediment contained slightly higher levels of metals for Hg, Pb, and Cd, and slightly higher concentrations of PCBs. After reducing the numbers to compare with actual turbidity levels due to bucket dredge disposal, however, the levels of toxicants were reduced to an insignificant fraction.

Salt marsh and mud flat succession was also discussed, with emphasis on the recent origin of most San Francisco Bay marshes due to gold mining hydraulic wastes.

124. Holmes, C.W., E.A. Slade, and C.J. McLerran.

1974. Migration and redistribution of zinc and cadmium in marine estuarine system. Environ. Sci. Tech. 8:255-259.

Heavy industrial discharges from a chemical plant lined harbor in the bay results in heavy release of Cd and Zn. In summer,

a stagnant, anoxic wedge is formed which is thickest in the harbor and declines into the bay. The metals are rapidly precipitated from this water into the sediments. Winter winds cause circulation to increase and bring metal poor, oxygen rich waters in over the bottom. In summer the oxygenated water over the anoxic wedge is metal rich. The winter metal poor water causes desorption from the metal rich sediment. Further in the harbor the water is forced to the surface and back out of the harbor. Up to 99% Zn and 79% Cd are sorbed onto suspended particulates in this water and transported back out to the bay where these sediments are redeposited. Winter water flow thereby works like a conveyor belt moving metals out into the bay long after they were deposited. Highly significant correlation was found with sulfides and authigenic phases of Zn in sediments and weaker correlation with Zn and E_h in water.

125. Hubbell, D.W., J.L. Glenn and H.H. Stevens, Jr.

1971. Studies of sediment transport in the Columbia River Estuary. p. 190-226, In, J.N. Nath and L.S. Slotta (eds.), Proceeding 1971 Technical Conference on Estuaries of the Pacific Northwest. Circular #42. Oregon State Univ., Corvallis, OR. 343 pp.

Information on sediment transport and deposition in the Columbia River estuary has been obtained by measuring and sampling the flow, surveying the bed with acoustic techniques, and determining radionuclide levels. Flow measurements and water-sediment samples show that temporal and spatial variations in suspended-sediment concentrations and in suspended-sediment discharges are large and are affected significantly by a turbidity maximum that develops and migrates longitudinally in the estuary. Side-scan sonar records obtained during a period of relatively low river flow indicate predominantly landward transport of sediment along the bottom in deep channels upstream from the mouth to about mile 14 and predominantly seaward transport on shallow slopes marginal to the channels downstream to about mile 5. A mass-balance equation that considers the amount of ^{65}Zn in the estuary bed and the net inflow of ^{65}Zn to the estuary suggests that approximately 30 percent of the silt and clay that enters the estuary from the river is retained there. Because of the complex character of estuary flows, detailed information on transport can be obtained only by making observations throughout tidal cycles and over extended periods of time.

126. Jernelöv, A.

1970. Release of methyl mercury from sediments with layers containing inorganic mercury at different depths. Limnol. Oceanogr. 15:958-960.

When no macroorganisms are present, deposits of inorganic mercury in the upper few centimeters of the sediment are responsible

for almost the whole amount of released methyl mercury. If Tubificidae or Anodonta are present at high densities, deposits down to 2.5 or 9 cm from the surface, respectively, are effective.

127. Jernelöv, A. and H. Lann.

1973. Studies in Sweden on feasibility of some methods for restoration of Hg contaminated bodies of water. Environ. Sci. Tech. 7:712-718.

Laboratory experiments and small-scale field tests have been performed to investigate some theoretically possible methods to restore mercury-contaminated lakes and rivers. Properties of the different methods with respect to practical use are discussed.

128. Krauskopf, K.B.

1956. Factors controlling the concentrations of thirteen rare elements in sea-water. Geochim. Cosmochim. Acta 9:1-32B.

The following factors controlling the concentrations of thirteen metals (Zn, Cu, Pb, Bi, Cd, Ni, Co, Hg, Ag, Cr, Mo, W, V) in seawater were studied: (1) precipitation of insoluble compounds with ions normally present in aerated sea-water, (2) precipitation of sulphides locally in reducing environments, (3) adsorption by ferrous sulphide, hydrated ferric oxide, hydrated manganese dioxide, apatite, clay, and organic matter. The precipitation reactions were investigated first by calculations based on solubility data, and then by experiments in which equilibrium concentrations of the metal ions were determined in sea-water in contact with their precipitates. Adsorption was measured by adding or precipitating the various adsorbents in sea-water samples containing an excess of one of the metal ions.

The principal conclusions are (1) Sea-water is greatly undersaturated with respect to all thirteen metals--in other words: precipitation of compounds with the ions normally present in aerated sea-water, even under extreme conditions of temperature and pH, cannot be responsible for the observed concentrations. (2) Local precipitation of sulphides is a possible control mechanism for seven of the elements (Cu, Zn, Hg, Ag, Cd, Bi, Pb), but is probably not the chief control because the concentrations are correlated to sulphide solubilities. (3) Adsorption is a possible mechanism for all elements except V, W, Ni, Co, and Cr: if Cr is assumed to be removed by local reduction and precipitation of the hydroxide, and the other four metals by organic reactions, the existing concentrations can be fairly adequately accounted for. Adsorption processes supplemented by organic reactions also furnish a qualitative explanation for the distribution of rare metals in sedimentary rocks of marine origin. These conclusions can probably be extended to other metals in the middle of the periodic system, but not to those on either side.

129. Lindberg, S.E. and R.C. Harriss.

1974. Mercury-organic matter associations in estuarine sediments and interstitial water. *Environ. Sci. Tech.* 8:459-462.

Sediment from the Florida Everglades and Mobile Bay estuary reveal significant associations between sediment Hg and sediment organic matter and between dissolved interstitial Hg and dissolved organic carbon. The bulk of dissolved Hg and dissolved organic carbon exists in the greater than 500 molecular weight fraction in Everglades pore water and in the less than 100,000 molecular weight fraction in Mobile Bay pore water. Mercury in sediments and interstitial water occurs at higher concentrations in the Everglades than in Mobile Bay, which receives anthropogenic mercury effluents. When normalized to organic content of the sediment or dissolved organic carbon concentration of the pore water, higher relative mercury concentrations occur in Mobile Bay. Interstitial dissolved mercury is enriched from 2.6 to 36 times over the associated surface water values, and in sulfide-rich pore waters far exceeds the thermodynamic solubility of HgS. Enrichment may be due to formation of organic and polysulfide complexes with mercury.

130. Loganathau, P. and R.G. Burau.

1973. Sorption of heavy metal ions by a hydrous manganese oxide. *Geochim. Cosmochim. Acta* 37:1277-1293.

Sorption of Co, Zn, Ca and Na by δ -MnO₂ was studied at $24.0 \pm 0.5^\circ\text{C}$ and pH 4. During the sorption of Co and Zn, Mn was released to the solution phase; however, Mn released was not detected during the sorption of Ca and Na. On the basis of crystal field theory, it is proposed that Zn may interchange with Mn²⁺ in the δ -MnO₂ structure, whereas Co may interchange with both Mn²⁺ and Mn³⁺. It is suggested that the interchangeable Mn²⁺ and Mn³⁺ sites were in the disordered layers in the δ -MnO₂ structure.

Sorption of Co, Zn and Ca at pH 4 fitted single-site Langmuir isotherm expressions at all Ca concentrations, but only at concentrations greater than 10^{-4} M for Co and Zn. Mn release by δ -MnO₂ at pH 4 during Co and Zn sorption also fitted single-site Langmuir isotherms. An expression for the case of multisite Langmuir sorption was derived and applied to the cases of Co and Zn sorption and to the case of Mn release during Co sorption. The data of those cases were used to calculate statistically the coefficients of multiple regression equations from which the sum of the capacities of all sites in each case were obtained. From all of these derived capacities it is proposed that there was only one site where Ca interchanged with surface bound H. Zn was postulated to interchange not only with these bound H sites, but also with another site where it interchanged with structural Mn²⁺. Co was postulated to interchange with both of these sites, and additionally with a third site where it interchanged with structural Mn³⁺. Using a pH-stat set at pH 4, it was determined that approximately 2 moles of H were released per mole of Co or Zn sorbed at bound H sites.

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131. Mortimer, C.H.

1971. Chemical exchanges between sediments and water in the Great Lakes--Speculations on probable regulatory mechanisms. *Limnol. Oceanogr.* 16:387-404.

As long as the oxygen concentration at the sediment surface does not fall below 1 or 2 mg/liter, typical deep-water sediments in the Great Lakes will probably be found to exert a measurable but quantitatively unimportant influence on the chemistry of the overlying waters. This conclusion is supported by the case of Windermere.

The sequence of more conspicuous changes that take place when the oxygen does fall below 1 mg/liter at the interface is illustrated by the case of Esthwaite Water, representative of events in lakes where biological production or organic pollution is high, where the subthermocline volume is relatively small, or both. A progressive decline in oxygen concentration from 2mg/liter to analytical zero at the interface was accompanied by a fall in electrode potential in the upper few millimeters of sediment, correlated with mobilization and transfer into the water first of manganese and later of iron. There is a concurrent transfer into the water of substantial quantities of phosphate, previously held in complex form, which may have important biological consequences. Other changes include liberation into the water of ammonia and silicate. Further reduction of the water-sediment systems permits microbial reduction of sulfate.

A third example of sediment-water exchange occurs during winter stratification under ice cover. In Torneträsk, in the course of 95 days of ice cover, a relatively thin layer of bottom water, initially at 1°C, gained heat and bicarbonate from and lost oxygen to the sediments. The density increase, arising from the heat gain, set density currents in motion that drained this contact water into the deepest parts of topographically isolated subbasins.

132. Moyer, B.R., and T.F. Budinger.

1974. Cadmium levels in the shoreline sediments of San Francisco Bay. LBL-2642, Prepared for the U.S. AEC, contract W-7405-ENG-48.

Cadmium levels in the low-tide shoreline sediments of San Francisco Bay were investigated at 68 locations. Sample cores of 5.5 cm diameter and 10 cm depth were separated into top and bottom fractions and analyzed for cadmium by atomic absorption spectrophotometry. A reliable and sensitive ion exchange/organic extraction technique was developed for analysis of cadmium to overcome the interference of iron and other metals that limit the usefulness of the conventional direct acid extraction technique.

The mean values of cadmium content in the top and bottom fractions (in dry soil) were 1.22 ± 0.99 (0.06 to 4.69) $\mu\text{g/g}$ and 0.93 ± 0.74 (0.14 to 3.91) respectively. The mean value for all

the 10-cm. cores of San Francisco Bay peripheral muds was $1.07 \pm 0.89 \mu\text{g/g}$. The top fraction cadmium content was 1.5 to 2 times more concentrated where the surface muds are better mixed; here the average cadmium content is 1.5 times that of the other sectors. This higher cadmium level is associated with the proximity of a lead smelter and several petroleum refineries. Concentrations greater than 2 ppm suggest recent pollution; 13% of the samples were above this level.

133. Nissenbaum, A., B.J. Presley and I.R. Kaplan.
1972. Early diagenesis in a reducing fjord, Saanich Inlet, British Columbia--I. Chemical and isotopic changes in major components of interstitial water. *Geochim. Cosmochim. Acta* 36:1007-1027.

Water and interstitial water from the reducing fjord of Saanich Inlet, British Columbia were analyzed for their major element composition, ammonia, phosphate, and silica contents, and for stable isotope composition of sulfur and carbon species. Ca was the only major element to show a significant change with depth (a 75 percent decrease in some cases). Ammonia and phosphate are highly enriched in the interstitial water (I.W.), concentrations reaching 250 ppm and 39 ppm, respectively. Total dissolved CO_2 in I.W. increases strongly with depth (20 to 30 times that in overlying sea-water) and it becomes enriched in C^{13} ($\delta\text{C}^{13} \approx +17.8 \text{‰}$). Both sulfate and dissolved sulfide decrease with depth to a complete disappearance of all sulfur species from the interstitial water. The dissolved sulfide is highly enriched in S^{34} ($\delta\text{S}^{34} \approx 18 \text{‰}$). All these changes are attributed to strong biological activity in the sediments.

134. Olausson, E.
1970. Water sediment exchange and recycling of pollutants through biochemical processes. 8 pp, In FAO Technical Conference on Marine Pollution and its Effects on Living Resources and Fishing. Rome, Italy, 9-18 December.

Man's effect on marine sediments is registered as an increased accumulation of carbonaceous matter with a change in pH and Eh of the environment, and in the burial of pollutants in the sediments. The redox discontinuity approached the water-sediment interface and in basins with restricted water exchange reached the sill depth due to eutrophication. A significant portion of the heavy metals (25 to 75%) seems to escape from the sediments into the water (organisms). The ratio PCB/ DDT changes rapidly with depth in sediments suggesting that PCB is not recycled while most DDT is recycled/decomposed rapidly.

135. Plumb. R.H. and G.F. Lee.

1973. A note on the iron-organic relationship in natural waters. *Wat. Res.* 7:581-585.

The percentage of iron in the aquatic environment which is associated with organic matter can be measured by fractionating unconcentrated water samples on Sephadex gel filtration columns. The use of concentrated samples should be avoided since they exaggerate the importance of the iron-organic relationship. Approximately 15-20% of Fe in Lake Mary is associated with organic matter. It appears to be held by acidic functional groups.

136. Presley, B.J., R.R. Brooks, and I.R. Kaplan.

1967. Manganese and related elements in the interstitial water of marine sediment. *Science* 158:906-910

Analyses for manganese, nickel, iron, cobalt, sodium, and lithium in the interstitial water of cores from the southern California borderland and six deep-sea cores in the area of the East Pacific Rise show great variation in concentration of trace elements. Oxidizing near-shore sediments showed a 50-fold enrichment in manganese in contrast to sulfide-rich reducing sediments, which showed no enrichment. Deep-sea sediments were variable in their concentration of the trace metals. All but one core showed a high enrichment in dissolved manganese, with a maximum of 6.6 parts per million. Two cores showed a 100-fold enrichment in nickel and cobalt. The manganese appears to be in solution either as Mn^{2+} or as a complex. The results appear to support manganese nodule formation in deep-sea sediments through a diffusion of manganese from depth to the surface.

137. Presley, B.J., J. Kolodny, A. Nissenbaum, and I.R. Kaplan.

1972. Early diagenesis in a reducing fjord, Saanich Inlet, British Columbia--II. Trace element distribution in interstitial water and sediments. *Geochim. Cosmochim. Acta* 36:1073-1090.

Sediment and interstitial water from four cores in Saanich Inlet were analyzed for total iron, manganese, nickel, cobalt, copper, molybdenum, zinc, lithium, and strontium. In addition, three separate sediment fractions were analyzed: (i) dilute acid soluble (ii) hydrogen peroxide soluble and (iii) non-extractable residue.

In the interstitial waters, Fe, Mn, and occasionally Zn showed enrichment varying from one to several orders of magnitude greater than the overlying sea water; the highest concentration was attained by Mn (3.3 ppm). Other elements were enriched by a factor of 2 to 5 above sea water concentrations. Enrichments are probably due to: (1) organic complexing of the metals and/or (2) equilibration with mineral phases which could generally not be identified in the solid.

Concentration of transition and base metals in the sediment is dependent on the source (provenance) of the detrital minerals and biogenic components (either skeletal tests or organic compounds). Little evidence was found for removal of these elements from the water column at the water-sediment interface. Diagenesis within the sediment column appears to be time-dependent. Mobilization of elements, particularly Cu, Fe, Ni, and Zn, takes place away from the insoluble silicate phase into the H_2O_2 soluble phase, which suggests organic complexing.

138. Sanchez, I. and G.F. Lee.

1973. Sorption of copper on Lake Monona sediments--effects of NTA on copper release from sediments. Wat. Res. 7:587-593.

A study has been conducted on the magnitude and mechanism of cupric binding to Lake Monona, Madison, Wisconsin, sediments. The binding capacity, which is the ability of the sediment to hold the copper, was found to be proportional to the carbonate content of the sediments. The addition of nitrilotriacetic acid to these sediments results in a release of iron and manganese and a reduction in the amount of copper present in solution.

139. Schutz, D.F. and K.K. Turekian.

1965. The investigation of the geographical and vertical distribution of several trace elements in sea water using neutron activation analysis. Geochim. Cosmochim. Acta 29:259-313.

Methods for the determination of 18 trace elements in sea water by neutron activation analysis have been developed. Of these elements sufficient analyses have been completed for gold, selenium, antimony, silver, cobalt, and nickel to permit a discussion of their distributions in the world ocean. The distributions of gold, selenium and antimony are more uniform than those of silver, cobalt, and nickel, so little can be said regarding the processes important to their supply to and removal from the world ocean. The variations observed for silver, cobalt, and nickel, however, make possible the evaluation of several factors which influence their concentrations in the ocean:

The Atlantic Ocean north of 10° S receives 60 percent of the total dissolved material supplied to the ocean by streams, but is lower in silver, cobalt and nickel than is the remainder of the world ocean indicating that continental run-off is not important in the deep-sea economy of these elements.

Relatively high concentrations of cobalt and nickel are found in water of the central Pacific Ocean in areas where the cobalt/manganese and nickel/manganese ratios in manganese nodules are high. The evidence that the nodule material in these areas is

of volcanic origin may also imply a volcanic origin for the cobalt and nickel in the overlying water. No such source is indicated for silver.

Unweathered rock material supplied directly to the ocean by Antarctic glaciers seems to be sufficient to affect the silica economy of the ocean and may in part affect the economy of other trace elements such as cobalt and nickel. The data are not sufficient, however, to permit an unambiguous interpretation.

Cobalt and nickel are relatively low in Long Island Sound compared to the North Atlantic indicating near-shore removal of these elements from sea water. In the deep sea much of the cobalt and nickel may be removed by co-precipitation with manganese oxide, but no mechanism seems singularly important in the removal of silver.

Increase in concentration of silver, cobalt, and nickel with depth in areas of high organic productivity indicates alteration of trace element concentration by organic reactions. In areas where upwelling currents oppose the downward movement of organic material, a relatively high steady state concentration may be produced.

140. Thomson, J., K.K. Turekian, and R.J. McCaffrey.

1974. The accumulation of metals in and release from sediments of Long Island Sound. 30 pp. In, K.K. Turekian (P.I.), Trace Elements in Natural Waters. Annual Progress Report to Atomic Energy Commission for Grant AT(11-1)-3573.

Sediment cores were taken from adjacent areas in Long Island Sound. One was cut into sections which were analysed for density, organic content, heavy metal content (Mn, Cu, Zn, Cd, and Pb), and radiochemistry (U^{238} , U^{234} , Th^{232} , and Th^{230}). One core was X-rayed for sedimentary structure. A gravity core, 76 cm. long, was analysed for Mn, Cu, Zn, Pb, Hg, and Pb^{210} . Some Ra^{226} measurements were also made. The results indicate that the shell rich zone in the upper 30 cm. (as shown by X-ray) has been accumulating for the last 70 years. Cu, Zn, Hg, and Pb data show that these metals increase from low values in the bottom 45 cm. to higher values (in the present time) in the top 30 cm. with the most increase in the top few cm. The Mn pattern is unique and suggests a possible migration up the sediment column. Uranium decreases from the bottom to the top of the core indicating a release of uranium in recent times as a result of oxidation. The rate of sediment accumulation from one core was 0.45 cm./yr. From this it was deduced that the major changes in metal supply rates began approximately 110 years ago. There is no evidence of remobilization of any of the trace metals. Even Mn which is mobilized in the reducing portion, probably does not escape the sediment column because of oxidation and precipitation as an oxide.

141. Zirino, A. and M.L. Healy.
1970. Inorganic zinc complex in seawater. *Limnol. Oceanogr.* 15:956-958.

The variation of zinc peak current with pH in the voltammetric analysis of seawater is interpreted in terms of a metal-ligand model. The model indicates that at the upper range of pH values in seawater, a substantial fraction of the zinc is present in complex forms. Preliminary field data are presented which suggest that these complexes may occur in the open ocean.

142. Zirino, A. and S. Yamamoto.
1972. A pH dependent model for the chemical speciation of copper, zinc, cadmium and lead in seawater. *Limnol. Oceanogr.* 17:661-671.

A pH-dependent model for the speciation of divalent Cu, Zn, Cd, and Pb ions in seawater was constructed with available and estimated thermodynamic stability constants and individual ion activity coefficients. This model was used to calculate the degree of interaction between each of the metal ions and the anions Cl^- , SO_4^{2-} , HCO_3^- , CO_3^{2-} , and OH^- as a function of pH. Interactions between a cation and an anion were assumed to result only in the formation of complexes with coordination of numbers of 1 to 4; polynuclear and mixed-ligand complexes were not included in the model.

The calculations showed the following: All four metals are complexed to a considerable extent in seawater; with the exception of Cd, the distributions of chemical species of the metals vary greatly with changes in pH; Cu interacts primarily with OH^- and CO_3^{2-} , Zn with OH^- , Pb with CO_3^{2-} and Cl^- , and Cd with Cl^- ; complexes with high coordination numbers (i.e. 3 and 4) are not formed to any appreciable extent in seawater.

E. INVERTEBRATE PHYSIOLOGY AND ECOLOGY

143. Adams, S.M. and J.W. Angelovic.

1970. Assimilation of detritus and its associated bacteria by three species of estuarine animals. Chesapeake Sci. 11:249-254.

The assimilation of detritus and its associated bacteria by three species of macrofauna which are abundant in eel grass beds during the summer was investigated in a laboratory study. The amount of $^{14}\text{CO}_2$ respired per milligram of dry body weight by animals that fed on different types of labeled food was compared to determine the relative amount of assimilation of a particular food. Animals were fed sterilized and unsterilized detritus labeled with ^{14}C to determine if they assimilated either the detritus per se or its associated bacteria as food.

All three species ingested and assimilated detritus. After three days of feeding, a small gastropod, Bittium varium, had ingested and assimilated more detritus per mg of body weight than either the grass shrimp, Palaemonetes pugio, or the polychaete, Glycera dibranchiata. Both Palaemonetes and Bittium assimilated ^{14}C from labeled detritus. All three species assimilated some ^{14}C from soluble sources.

144. Anderson, A.E., E.C. Jonas, and H.T. Odum.

1958. Alteration of clay minerals by digestive processes of marine organisms. Science 127:190-191.

Experiments were conducted with molluscs, echinoderms, filter feeding fish, and plankton to determine the effects of digestion on clay particles. Of the three materials used (kaolinite, bentonite, and interlayered soil clay), the kaolinite was not affected by any of the organisms or sea water. However, the bentonite and the interlayered clay exhibited appreciable changes in both the controls (sea water) and the organisms after 24 hrs. and 5 days. These clays, upon exposure to seawater, began to take up magnesium immediately and gradually began development of a brucite configuration between mica layers. Similar clays which pass through organisms, however, lose brucite and build up potassium as an interlayered cation.

145. Bader, R.G.

1954. The role of organic matter in determining the distribution of pelecypods in marine sediments. J. Mar. Res. 13:32-47.

Sediments in the region of Mt. Desert Island, Maine, indicate that their organic content and its state of decomposition, as expressed by $L/R_n \times T.O.$ (where L = refractory organics,

R_n = nonrefractory organics, and T.O. = total organics), are primary factors in controlling the distribution of sediment-dwelling pelecypods. The relationship of pelecypod population density to the physical characteristics of the sediments and the depth of overlying water are apparently secondary in importance.

146. Black, R.

1973. Growth rates of intertidal molluscs as indicators of effects of unexpected incidents of pollution. J. Fish Res. Bd. Canada 30:1385-1388.

Shells of living molluscs contain a record of previous growth rate if the time interval between check marks is known. The check marks in Mytilus edulis and Littorina littorea are probably annual marks. Growth rates of Mytilus in 1969, the year of significant elemental phosphorus pollution at Long Harbour, Newfoundland, were no different from those in preceding or succeeding years. No Littorina alive in 1969 occurred at Long Harbour, but they were abundant at other locations. This sort of analysis, used with caution because effects of pollutants may be confounded with effects of other variables, provides a method of examining nonlethal effects of unexpected polluting events.

147. Bohle, B.

1972. Effects of adaptation to reduced salinity on filtration activity and growth of mussels (Mytilus edulis). J. Exp. Mar. Bio. Ecol. 10:41-47.

The filtration activity and growth of juvenile mussels adapted to 100% sea water and placed in 100, 75, and 50% sea water have been studied. The mussels in 50% sea water required 8-9 weeks to reach the activity level of 100% mussels while at 75% the mussels required 4 weeks. Only the mussels previously adapted in 100% sea water showed a significant length increase. This implies that mussels living in the sea with strongly varying salinity have reduced growth rate.

148. Boyden, C.R. and C. Little.

1973. Faunal distributions in soft sediments of the Severn Estuary. Estuar. Coast. Mar. Sci. 1:203-223.

The pattern of the infauna on particulate shores of the southern coast of the Severn Estuary has been examined. Although mud predominates as the major beach substratum within the Estuary, sand beaches occur in both the lower and upper reaches. Liquid mud is found at l.w. in the middle reaches. Suitability of substratum is shown to be the most important factor influencing animal distribution. The number in infaunal species found in this survey did not increase markedly towards the mouth, in contrast to the pattern

displayed by rocky shore animals. Several species were rare or absent on mud beaches close to the entrance of the River Avon and the possibility that sewage contamination is responsible for this is discussed.

149. Bubnova, N.P.

1972. Feeding of the detritus-feeding molluscs Macoma baltica (L) and Portlandia artica (Gray) and their influence on bottom sediments. Okeanologiya 12:1084-1090. (Russian).

The amount of sediment reworked by molluscs during feeding per day is 10 times as large as the rations of Macoma and only twice as large as the rations of Portlandia. Macoma assimilates 21 percent of the dry matter, 70 percent of the organic matter and 62 percent of the albumen. The assimilability of dry and organic matter and of albumen for Portlandia is 35 percent, 50 percent and 51 percent respectively. The population of M. baltica in the littoral zone reworks and consumes more sediment than Portlandia in the sublittoral zone.

150. Castagna, M. and P. Chanley.

1973. Salinity tolerance of some marine bivalves from inshore and estuarine environments in Virginia waters on the Western Mid-Atlantic coast. Malacologia 12:47-96.

Many species of estuarine bivalves have a distribution pattern closely correlated with salinity, indicating the importance of salinity in determining these patterns. The approximate salinity tolerance range for 36 species of bivalves is described. Tolerance limits for 29 species were determined in laboratory experiments. Most of these species display a remarkable degree of euryhalinity. All survived a minimum salinity of at least 17.5‰ and 25 species survived at 12.5‰. Twenty species survived at various lower salinities.

Salinity tolerance for a given species is not constant but varies with season, salinity experience, and temperature. Burrowing, feeding, and reproduction usually occur at nearly all salinities at which survival is possible. Byssal formation requires a higher salinity than is necessary for other activities.

In Virginia about two-thirds of the species of salt-water bivalves discussed can be found over the entire salinity range they are capable of tolerating in the laboratory. Eleven species do not occur over their entire potential salinity range. Eight of the 11 species (Yoldia limatula, Mytilus edulis, Venericardia tridentata, Lucina multilineata, Dosmia discus, Abra aequalis, Mya arenaria, Martesia cuneiformis) are near the geographic limit of their range; their distribution locally may be limited primarily by the factors that determine their geographic range. The distribution of 5 species,

Argopecten irradians, Conger leucophaeta, Macoma mitchelli, Donax variabilis, and Spisula solidissima) may be influenced by predation, competition, or special environmental requirements. Four of the 11 species (Conger leucophaeta, Macoma mitchelli, Donax variabilis, Rangia cuneata) occur in specialized habitats with low species diversity.

151. Chapman, G. and A.G. Taylor.

1968. Uptake of organic solutes by Nereis virens.
Nature 217:763-764.

Nereis virens was used in an experiment concerned with uptake of glutamic acid from solution. Some worms were free in the solution, while others were ligatured at both ends. Results were similar for both. Uptake occurred, but it was not directly proportional to concentration, indicating that the process is probably not controlled by simple diffusion. Since these organisms can take up a range of organic compounds from solution, this process is likely to be significant in their overall feeding economy.

152. Chiba, K. and Y. Ohshima.

1957. Effect of suspended particulates on the pumping and feeding of marine bivalves, especially of the Japanese little neck clam. Bull. Jap. Soc. Sci. Fish. 23: 348-353.

In natural condition of sea water there are many grades of turbidity originated from varied kinds of particles, and it is important to know the effect of suspended particulates on the living processes of bivalves in our technique of shell culture. Some experiments were carried out to test the rate of pumping and feeding of bivalves (Venerupis semidecussata, Mercenaria meretrix, Ostrea gigas and Mytilus edulis) by means of high concentration of particles.

The rate of pumping was measured by using the formula of Jorgensen and that of feeding by weighing the dried feces excreted by the bivalves after an hour in sea water to which bentonite had been added.

The results are summarized as follows: (1) In each species of organisms tested the rate of pumping was not reduced due to the high concentration of bentonite. (2) In V. semidecussata the amount of feeding increased with the growing concentration of the particles up to the time when pseudo-feces were discharged, and the pseudo-feces increased further, whereas the true feces did not increase in the higher concentrations. Nearly the same tendency of feeding was observed in M. meretrix. (3) It was found in O. gigas and M. edulis that the amount of feeding increased with the growing concentration of bentonite, but the rate of its increment grew smaller.

153. Dales, R.P.

1955. Feeding and digestion in terebellid polychaetes.
J. Mar Biol. Ass. U.K. 34:55-79.

The feeding and the functional morphology of the gut of Amphitrite johnstoni are described, and later compared with that of other terebellids. Feeding is effected by the tentacles bringing particles to the lips, where some sorting occurs, and the control of movement of the tentacles and the lip mechanism are described. The gut in all terebellids consists of an oesophagus, fore-stomach (comprising 10-15% of the total gut-length), muscular hind-stomach (5-12%) and intestine (65-75%). The muscular hind-stomach acts as a mixer and contains a peritrophic membrane. Enzymes are secreted by the fore-stomach and the fore-intestine. Strong amylase activity, weak protease activity and lipase activity is found in the fore-stomach; strong protease activity, amylase and lipase activity is found in the fore-intestine, and these differences are correlated with differences in histology. The gut has a pH of 6.0 in the digestive regions in feeding worms. Absorption occurs in the fore-intestine and the anterior part of the hind-intestine. Transmission of food throughout the gut is caused mainly by muscular action. The lip structure of Amphitrite, Polycirrus, Lanice, Terebellides and Artacama is briefly described in relation to the different habits of these genera.

154. Davids, C.

1964. The influence of suspensions of microorganisms of different concentrations on the pumping and retention rate of food by the mussel (Mytilus edulis). Neth. J. Sea Res. 2:233-249.

This study of Mytilus edulis is to determine the influence of food concentration on the amount of water pumped and the amount of food retained. It was found that the amount of water pumped is not always the same. For a mussel of length 2.9 cm it varied between 0.35 and 1.05 l per hour. At times pumping would stop for a short time. In very dilute suspensions pumping was faster than in seawater. In general, the rate of pumping depends on the concentration of the previous suspension. After very concentrated suspensions, the pumping rate will be much greater in lower suspensions and seawater. For the algae Nitzschia and Isochrysis, the retention rate was very high. However, Chlorella were very poorly retained, leading to the conclusion that they are not an acceptable food for Mytilus. No obvious connection was found between the concentrations of the suspension and the retention percentage, but in a concentration of over 400,000 cells/ml, ejection of pseudo-faeces occurs.

155. Davis, C.C.

1970. The effects of pollutants on the reproduction of marine organisms. 13 pp. In, FAO Technical Conference on Marine Pollution and Its Effects on Living Resources and Fishing. Rome, Italy, 9-18 December.

The effects of different pollutions on reproductivity are reviewed, including: thermal pollution, salinity changes, insecticides, oil and detergent pollutants, inorganic pollutants, radioactive wastes, and eutrophication effects. Research needs are discussed, citing need for both laboratory and field work in this area.

156. Driscoll, E.G. and D.E. Brandon.

1973. Mollusc-sediment relationships in northwestern Buzzards Bay, Massachusetts, U.S.A. *Malacologia* 12:13-46.

Four facies are defined in the recent sediments of the north-western part of Buzzards Bay, Massachusetts. These are characterized by differences in mean grain diameter, sorting, silt-clay content and fauna. A mean grain diameter of less than 2.5 ϕ (0.18 mm) is characteristic of three of these facies which are found in protected areas. The fourth, developing on current-swept bottoms, typically is composed of coarser sediments. Calcium carbonate content of the sediment is a reflection of dead shell abundance throughout the area.

Faunas of 3 fine-grained facies are characterized respectively by (1) Nucula proxima, (2) Yoldia limatula and Nassarius trivittatus and (3) Macoma tenta, Nucula proxima, Eupleura caudata, and Nassarius trivittatus. No more than 8 species compose 1% or more of the molluscan fauna in any of the 3 fine-grained facies. In the coarser sediments of the 4th facies, 11 species comprise in excess of 1% of the molluscs. Dominant species in this facies are Nassarius trivittatus, Anachis avara similis, Chaetopleura apiculata, Anadara transversa and Crepidula fornicata.

The distribution of more than 35 molluscan species is discussed. The majority have hard parts which are potentially preservable. Mean grain diameter, abundance of silt and clay, and presence of dead shells are important factors in mollusc distribution. Macoma tenta, Yoldia limatula, Nucula proxima and Solemya velum increase in abundance with decreasing mean grain diameter and increasing silt-clay content of the sediment. Chaetopleura apiculata, Anadara transversa, Crassinella mactracea and many other species increase in abundance with increasing mean grain diameter and decreasing silt-clay content of the sediment. Greater abundance and diversity of epifaunal species is found on bottoms having higher concentrations of dead shells.

Potential faunal-lithic associations indicate that suspension feeding bivalves and carnivorous gastropods are available

for preservation in the coarse-grained facies. Potential fossils of the proto-graywacke, which makes up the 3 fine-grained facies, are mostly deposit feeding bivalves.

157. Dral, A.D.G.

1967. The movements of the laterofrontal cilia and the mechanism of particle retention in the mussel (Mytilus edulis L.) Neth. J. Sea Res. 3:392-422.

This article describes methods whereby cilia on intact gills can be studied using transparent young mussels. A full description of the movement of the latero-frontal cilia is given. The beat frequency varies and is affected by the resistance in the water current; food particles stick to the latero-frontal cilia. In high concentrations of suspended matter only partial retention of particles was observed, and in some cases of extreme concentration pumping would stop altogether. At low pumping rates all particles of 3 μ and up are retained, whereas the particle retention at higher pumping rates is 6 μ or larger.

158. Eagle, R.A. and E.I.S. Rees.

1973. Indicator species: a case for caution. Mar. Poll. Bull. 4:25.

Capitella capitata has often been found around sewage outfalls in large numbers and is therefore considered a good indicator of high organic enrichment. However, in the case of an outfall in Liverpool Bay, large numbers of C. capitata were found before the pipe had actually gone into use and disappeared 12 months later. This led the authors to conclude that the disturbance of the area was responsible rather than the effluent.

159. George, J.D.

1964. Organic matter available to the polychaete Cirriformia tentaculata (Montague) living in an intertidal mud flat. Limnol. Oceanogr. 9:453-455.

Experiments were conducted with Cirriformia tentaculata to determine the amount of organic matter available to the non-selective surface deposit feeder. Samples of the top 0.5 cm of mud were taken and particles < 0.4 mm were analyzed for organic content. 0.4 mm is the largest particle size ingested by C. tentaculata. These samples were digested with enzymes from the gut of the worm in order to estimate the amount of organic material available for assimilation, and organic carbon contents of the enzyme treated mud were compared with amounts in the feces. From the digestion experiment, it was found that 14.5% of the organic matter available for feeding was digested by the enzymes. The carbon content of the feces was lower than the digested mud. Only 7.9% of the available organics was actually digested by the animal.

160. Gilbert, M.A.

1973. Growth rate, longevity, and maximum size of Macoma balthica (L.). Biol. Bull. 145:119-126.

Sixty specimens of M. balthica were tagged and measured (to nearest 0.1 mm), then put in a wooden box filled with native sediment and placed into Rand Harbor. The box was removed four times during the year, and the tagged animals were measured and replaced. Dry weight biomass, dry shell weight, and caloric content were also determined.

M. balthica were found to grow all year round with the greatest growth occurring from May to August. Most of the length is achieved in the first 2 years of life and slows down near 22-25 mm. Average longevity is estimated at 6-10 years.

The results of this study was compared with others and it was suggested that size and growth rate decrease and longevity increases with increasing latitude.

161. Gordon, D.C. Jr.

1966. The effects of the deposit feeding polychaete, Pectinaria gouldii on the intertidal sediments of Barnstable Harbor. Limnol. Oceanogr. 11:327-332.

Under laboratory conditions, individual Pectinaria gouldii worked 6 g of sediment/day. This rate was related directly to sediment temperature and inversely to sediment pigment concentration. At this rate, corrected for the effects of daily tidal exposure and annual temperature fluctuation, the worm population in the flat studied (10 worms/m²) would completely work all sediment in the 6-cm thick surface layer every 15 years. The observed activity of other deposit feeders would substantially reduce this time. P. gouldii removed 3.55 units of sediment pigment (42%) and 1.43 mg of organic carbon (45%) from each gram of sediment worked, suggesting a rapid utilization of the available organic matter.

162. Matthiessen, G.C

1960. Observations on the ecology of the soft clam, Mya arenaria, in a salt pond. Limnol. Oceanogr. 5:291-300.

Observations were made on the ecology of the soft clam, Mya arenaria, in a salt water pond on Martha's Vineyard, Mass., during the summer and fall of 1958. Primary consideration was given to the tolerance of this species to low and fluctuating salinities, the effect of salinity upon rate of growth, and possible correlation between growth rate and the nature and abundance of phytoplankton.

Field observations supplemented by laboratory experimentation indicated that Mya longer than 2 mm can tolerate salinities less than 1‰ for at least 24 hrs. and can withstand sudden salinity

fluctuations of an order of 18%. Tolerance of clams 2-25 mm in length to low salinities was found to be proportional to their size.

Growth rate of clams living under different salinity conditions ranging in mean value from 3.5‰ to 14‰ was found to vary directly with salinity. The fact that the pumping rate of clams was observed to vary directly with salinity within the limits 0-31‰ is considered significant in this respect.

Comparisons of growth rates with phytoplankton concentrations suggested that flagellates form an important source of nutrition for Mya and that diatoms may be of relatively little importance.

163. Maurer, D.

1967. Mode of feeding and diet, and synthesis of studies on marine pelecypods from Tomales Bay, California. *Veliger* 10:72-76.

The main object of this study was to obtain information concerning response of molluscs to sediment in terms of food, protection, larval sites, and turbidity. It appears that sediment as a food source may dictate mode of feeding, and that certain conditions of turbidity are related to deposition of particular sediment types. It was also found that turbid conditions inhibited growth and clean environments fostered it in the pelecypods studied.

164. Newell, R.

1965. The role of detritus in the nutrition of two marine deposit feeders, the prosobranch Hydrobia ulvae and the bivalve Macoma balthica. *Proc. Zoo. Soc. London* 144:25-45.

Observations on the shore show that the density of the population of the prosobranch Hydrobia ulvae and of the bivalve Macoma balthica at Whitstable varies considerably with the grade of deposit, dense populations being found in fine deposits and sparse populations in coarse deposits and that the micro-organisms are more abundant in fine deposits than in coarse ones.

As a result of these conclusions it is suggested that the rapid increase in population density of Hydrobia ulvae and Macoma balthica, and, by inference, of other deposit-feeders, towards regions of fine-grained deposits is attributable to the increase in density of the micro-organism population. This, in turn, is related to the surface area of the deposits rather than to the abundance of organic debris.

Finally, the nature of detritus is discussed and it is recommended that this term is replaced by "organic debris".

165. Peeters, J.C.H. and W.J. Wolff.

1973. Macrobenthos and fishes of the rivers Meuse and Rhine, Netherlands. *Hydrobiol. Bull.* 7:121-126.

The benthic fauna clearly indicates that the water of the Waal is strongly polluted. The characteristic fluviatile fauna has completely disappeared and only a few ubiquitous maintain themselves in low numbers. Only the tubificid oligochaetes, well known indicators of pollution, occur in large numbers.

The benthic fauna of the Meuse, on the other hand, shows a much richer picture with even a number of characteristic riverine species. Moreover, a few species characteristic of stagnant waters, e.g. Limnaea auricularia and Physa acuta, occur in the part of the river where stows impede the water flow. Although the occurrence of oligochaetes points to a certain pollution, the overall picture shows a much better water quality in the Meuse than in the Waal.

The fish faunas of the Waal and the Meuse only differ with respect to gudgeon, occurring more in the Waal, and to bream, pike and pike-perch occurring more in the Meuse. However, these data do not lead to the conclusion that the number of fish species occurring in the Meuse is higher than in the Waal. Compared with the large differences found in the benthos, this result is unexpected.

166. Rhoads, D.C.

1973. The influence of deposit-feeding benthos on water turbidity and nutrient recycling. *Amer. J. Sci.* 273:1-22.

The central axis of Buzzards Bay, Mass., is characterized by silt-clay muds populated by deposit-feeding bivalves and polychaetes. This facies lies generally below a depth of 13 m. The concentration of suspended solids (turbidity) is greatest over this bottom type throughout the bay. The areal distribution of this turbidity zone and its relationship to bottom properties, fauna, and tidal flow have been mapped and studied over an area of 20 km². This type of bottom extends over a total area of 160 km² in the bay.

The concentration of suspended solids increases with depth, reaching maximum values (10-35 mg/l) within 3 m of the bottom. The lateral boundaries of the turbidity zone are coincident with the limits of intensive deposit-feeder activity. Maximum turbidity appears on the ebb phase of the tidal cycle.

Suspended seston is derived from resuspension of bottom sediments. Intensive biologic reworking of the bottom makes this surface physically unstable in the presence of weak tidal flow. Many of the suspended particles are identifiable as fecal pellets of deposit-feeding species.

The potential importance of the permanent "upwelling"

of the bottom for molluscan aquaculture has been explored by transplanting oysters (Crassostrea virginica) into this turbidity zone on metal racks which support the bivalves above the bottom. Initial results from Long Island Sound indicate higher mean meat yields in oysters feeding in this turbidity zone than in oysters grown on commercial beds in shallow water. The nutrient value of the suspended seston has not yet been demonstrated.

167. Reish, D.J.

1955. The relation of polychaetous annelids to harbor pollution. Public Health Rep. 70:1168-1173.

A physical, chemical, and biological study of 55 sampling stations made 3 times during 1954 in Los Angeles-Long Beach Harbors, Calif., placed emphasis upon bottom organisms, particularly the polychaetous annelids. As a result of the survey, the harbors can be divided into five zones on the basis of bottom conditions and distribution of polychaetes. The zones are: (1) A healthy bottom area characterized by a large number of polychaete species, with Tharyx parvus, Cossura sp., and Nereis procera predominating. (2) A semihealthy bottom area, with Polydora (Carazzia) paucibranchiata and Dorvella articulata predominating. (3) A second semihealthy bottom area in which Cirriiformia luxuriosa predominates. (4) A polluted bottom area, with Capitella capitata predominating. (5) A very polluted bottom area in which there is no animal life.

168. Stephens, G.C.

1967. Dissolved organic material as a nutritional source for marine and estuarine invertebrates. P. 367-375 In, G.H. Sauff (ed.), Estuaries. The Horn-Shafer Co. Baltimore, MD.

This paper is a summary of recent evidence concerning a possible nutritive role of organic material in true solution.

The uptake of organic compounds of low molecular weight appears to be a significant supplemental source of reduced carbon for several marine invertebrates. Comparison of the material available to the animals at concentrations characteristic of their environment and the amount of material necessary to account for oxidative metabolism supports this position. There is also qualitative evidence that such material can enter oxidative and synthetic pathways in the animals studied. There is no intent to deny the significance of other means by which marine invertebrates obtain their food. For example, a detritus feeder indeed passes material through the gut. It is proposed, however, that the uptake of organic material from solution may supplement other pathways.

169. Tenore, K.R., J.C. Goldman, and J.P. Clarnier.

1973. The food chain dynamics of the oyster, clam, and mussel in an aquaculture food chain. J. Exp. Mar. Biol. Ecol. 12:157-165.

The food chain dynamics of the edible mussel Mytilus edulis L., the American oyster Crassostrea virginica (Gmelin) and the hard clam Mercenaria mercenaria (L.) were investigated in large experimental tanks with flowing, filtered sea water and controlled addition of phytoplankton. The feeding rate of the mussel (5.36 μ g carbon removed/1/g C per animal) was higher than that of the oyster (3.92) and clam (3.03) but the ecological efficiencies (net production/ingested food) X 100 of the clam (23.69%) and the oyster (18.38%) were higher than that of the mussel (10.01%).

The food chain efficiencies (net production/available food) were lower than the ecological efficiencies, suggesting under-exploitation of the available food. The clam, although having a lower feeding rate, was more efficient in utilizing the food it filtered and so showed the highest net production.

The rates (μ g-at/1/g C animal) of regeneration of nutrients, especially total inorganic nitrogen (mussel, 2.1723×10^{-3} oyster, 7.4270×10^{-3} ; and clam, 8.1750×10^{-3}) along with reported high biodeposition rates of bivalves suggest that multi-species aquaculture systems would be more efficient and productive than one-species systems.

170. Tenore, K.R. and K.M. Dunstein.

1973. Comparison of feeding and biodeposition of three bivalves at different food levels. Mar. Biol. 21:190-195.

Experiments on the edible mussel Mytilus edulis, the American oyster Crassostrea virginica, and the hard clam Mercenaria mercenaria, using flowing systems, showed that the feeding and biodeposition rates were affected by food concentrations. At all levels of food concentration, the order of increasing feeding rate (both the percent of available particulate carbon and the actual amount of carbon removed) was: clam < oyster < mussel. All bivalves exhibited lower feeding rates (both percent and actual) at low food concentrations. However, the percent of available food removed quickly increased to a maximum at food concentrations typical for the natural environment. This maximum remained constant for the mussel and oyster, but declined with increasing food concentration for the clam. However, because this percentage was for increasing levels, the actual carbon removed continued to increase up to the highest food level for all three bivalves. In increasing order of biodeposition rate, the bivalves were: clam < oyster < mussel. The biodeposition rates of the three bivalves increased logarithmically with increased food concentration as a result of the production of pseudofeces. The feeding and biodeposition data were used to calculate assimilation rates, and this pointed out the higher efficiency of the oyster compared to the mussel and clam.

171. Theisen, B.F.

1972. Shell cleaning and deposit feeding in Mytilus edulis L. (Bivalvia). Ophelia 10:49-55.

Mytilus edulis as a rule has shells with a clean and smooth periostracum. It is shown that the mussels actively clean their shells with the foot, which is morphologically adapted for the cleaning process. Particles removed from the shells during cleaning are taken into the mantle cavity and treated in the same way as particles strained from the water by the gills, i.e. eaten or rejected as pseudofaeces. Particles reached by the foot during creeping may be treated in the same way.

172. Vahl, O.

1973. Pumping and oxygen consumption rates of Mytilus edulis L of different sizes. Ophelia 12:45-52.

Specimens of M. edulis were collected in April of 1972 in Helsingør, Denmark in water of 15.5‰ salinity and 12.2°C. They were acclimated for 30 days at a salinity of 21-23‰ and temp. of $10 \pm .5^\circ\text{C}$ where they were fed a mixture of Isochrysis galbana and Monochrysis lutheri. For the pumping experiments, the concentration of flagellates ranged from 2000-6000 cells/ml and the size ranged from 4.7-7.3 μ . This size (4.8-7.3 μ) was used because they are 100% retained on the Mytilus gill.

The pumping rate, O_2 consumption, and absorption efficiency were measured and compared to gill area. The ratios were transformed to radians and treated according to Sokal and Rohlf.

It was discovered that the gill area and metabolism increase faster with size than the pumping rate. The pumping rate to O_2 uptake (P/R) shows that the energy available for growth and reproduction decreases with decreasing size. Absorption does not change significantly with size. Compared to some other invertebrates, it was concluded that M. edulis has a higher energy profit per calories spent due to higher pumping rate per m. O_2 consumed and a higher particle retention rate.

173. Whitlatch, R.B.

1974. Food-resource partitioning in the deposit feeding Pectinaria gouldii. Biol. Bull. 147:227-235.

A study of the food-resource partitioning in the deposit-feeding polychaete Pectinaria gouldii collected from Little Sippe-wisset salt marsh, Massachusetts, shows that, on the average, larger worms select larger particles than smaller worms. Comparisons of ingested sediment with sediment collected where the animals were feeding indicate that the polychaetes prefer organic-encrusted mineral grains, floc aggregates, and fecal material.

Histological stains were used to determine the percentage particle abundance of different possible food sources and fractions

ingested by the polychaetes. Mercuric bromphenol blue (MBB) was used to stain protein-containing material and periodic acid Schiff reagent (PAS) was used to stain carbohydrate-protein complexes. Total possible organic material in the sediment averaged 32.7%. Very little of the sediment (less than 0.4%) stained with MBB, while an average of 13.9% of the sediment stained with PAS. Of the total possible organic matter, only about one-half stained with the PAS reagent suggesting not all of the material is organic in nature.

Analysis of the sediment ingested by the worms averaged 42.7% possible organic matter, of which 20.5% was PAS-stained. Calculations of the assimilation efficiencies of P. gouldii show that the worms remove, on the average, 30% of the possible organic matter and 29.1% of the stained material from the sediment.

174. Winter, J.E.

1969. On the influence of food concentration and other factors on filtration rate and food utilization in the mussels Artica islandica and Modiolus modiolus. Mar. Biol. 4:87-135.

Filtration rates and the extent of phagocytosed food particles were determined in the offshore lamellibranchs Artica islandica and Modiolus modiolus in relation to particle concentration, body size and temperature. Pure cultures of the algae Chlamydomonas sp. and Dunaliella sp. were used as food.

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